An Abstract of the Thesis of

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Abstract approved: ______________

James A. Moore

Five unique but related studies were conducted at the Oregon State University Dairy Wetland Treatment System (OSUDWTS), Corvallis, OR. The research site consisted of six parallel wetland cells, which were built in 1992 and began receiving concentrated dairy wastewater in the fall of 1993. Hydrologic, hydraulic, and water quality data were collected at the site for three years. The five resulting studies were:

1. the prediction of evapotranspiration (ET) from wetlands;
2. the development of a hydrologic model and water budget for the OSUDWTS;
3. a preliminary investigation of the hydraulics of the OSUDWTS;
4. an overall evaluation of the treatment performance of the OSUDWTS and applicability of current constructed wetland design methods to livestock wastewater wetlands; and
5. the development of a conceptual model for nitrogen removal in constructed wetlands.

Average ET rates for the wetland cells were found to be 1.6 times as great as the Penman-Monteith alfalfa reference ET. Specific crop coefficients were 1.72, 2.32, and 0.57 for bulrush, cattails, and floating grass mats. The detailed hydrology model predicted daily

water levels very accurately ($R^2 = 0.95$) and showed seasonal rainfall and ET could increase or decrease the average detention time by as much as 18%.

Tracer studies indicated that non ideal flow existed in the wetlands. Actual detention times were found to be an average of 43% shorter than theoretical detention times. Tank-in-series and plug flow modified by dispersion models were inadequate at describing the observed tracer response.

Constructed wetlands were shown to be able to reduce a high percentage of most waste constituents in concentrated livestock wastewaters. Average reductions for COD, BOD, TS, TSS, TP, TKN, NH_3 and fecal coliforms were 45, 52, 27, 55, 42, 41, 37 and 80%, respectively. Rate constants for volumetric and areal first-order plug flow models were found for each wastewater constituent. Overall, both models were fair at predicting wastewater reduction at the OSUDWTS.

A conceptual model of nitrogen cycling showed denitrification to be the most important process for nitrogen removal in constructed wetlands. However, low dissolved oxygen in constructed wetlands limits nitrification, which in turn limits denitrification.

Treatment of Dairy Wastewater in a Constructed Wetland System: Evapotranspiration, Hydrology, Hydraulics, Treatment Performance, and Nitrogen Cycling Processes

by

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Steven F. Niswander, Author

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Treatment of Dairy Wastewater in a Constructed Wetland System: Evapotranspiration, Hydrology, Hydraulics, Treatment Performance, and Nitrogen Cycling Processes

1. Introduction

The use of constructed wetlands has increased dramatically in recent years. One of the latest uses has been for treatment of concentrated livestock wastewater. Preliminary results indicate potentially high removal of biochemical oxygen demand, total suspended solids, total nitrogen, total phosphorus, and fecal coliforms (Knight and Kadlec, 1996). However, performance data are highly variable and only simple empirical equations exist to predict treatment. If constructed wetlands are going to be accepted as a treatment technology for livestock wastewater, then reliable design and removal equations must exist.

Treatment in constructed wetlands is influenced by several factors including contacting pattern and time. It is generally assumed that the flow in constructed wetlands can be described as plug flow, which means all water spends an equal amount of time in the wetland. The time of treatment is simply the volume of the wetland divided by the inlet flow rate. This value is theoretical detention time. The most common equation used to predict treatment in constructed wetlands is a first-order irreversible reaction:

$$C_o = C_{in} \cdot e^{-k_t \cdot t}$$

where, C_o = concentration out (mg/l),

C_{in} = concentration in (mg/l),

k_t = temperature dependent first order rate constant (d^{-1}), and

t = time (d).

This has been shown to be accurate for many wastewater constituents (Kadlec and Knight, 1996; Reed et al., 1995), however several factors influence the actual detention time. These include water losses and gains (caused by precipitation, evapotranspiration and infiltration), non-uniform flow, and the volume occupied by plant material and litter.

The objectives of this study are to:

1. evaluate current methods for calculating evapotranspiration from wetlands;
2. predict daily evapotranspiration from the Oregon State University Dairy Wetland Treatment System (OSUDWTS);
3. develop a complete water budget for OSUDWTS;
4. determine the hydraulic characteristics of the wetland cells at OSUDWTS;
5. measure the treatment performance of OSUDWTS;
6. evaluate current design equations for constructed wetlands treating livestock wastewater; and
7. review nitrogen cycling processes that influence nitrogen treatment in constructed wetlands.

2. Evapotranspiration from a Constructed Wetland System containing *Thypha latifolia*, *Scirpus acutus*, and floating grass mats (*Glyceria occidentallis* and *Alopercus geniculatus*)

2.1 Abstract

Evapotranspiration (ET) coefficients were developed for bulrush (*Scirpus acutus*), cattail (*Thypha latifolia*), and floating grass mats (Western mannagrass (*Glyceria occidentallis*) and water foxtail (*Alopercus geniculatus*)) in six 28.1 m x 5.9 m x 0.3 m wetland treatment cells located in Corvallis, OR. Daily water loss measurements and an annual water budget for 1996 were used to fit segmented crop coefficient curves to Penman-Monteith alfalfa reference evapotranspiration. Maximum ET rates were 1.72, 2.32, and 0.57 times greater than reference ET for bulrush, cattails, and floating grass mats, respectively. Average ET for all wetland cells was 1.60 times greater than reference ET. An overall water budget for the system indicated that the ET calculations were accurate throughout the year ($R^2 = 0.95$).

Keywords: energy balance; evaporation; transpiration; Penman-Monteith

2.2 Introduction

Evapotranspiration (ET), also referred to as consumptive use, is the transport of water to the atmosphere by the process of evaporation and transpiration. Evaporation is caused by vaporization of water from water and soil surfaces. Transpiration is the passing of water through vascular plants to the atmosphere. ET is defined as "the quantity of water transpired by plants during their growth or retained in the plant tissue, plus the moisture evaporated from the surface of the soil and the vegetation" (ASCE, 1949).

ET plays a very important role in the water budget of wetlands. It is critical that methods exist for accurately predicting ET from wetlands as more demand is put on existing water supplies and more emphasis is placed on watershed management. Accurate ET measurements are also needed for calculating water use by wetlands. This is important for both the management of natural wetlands, where water is often withdrawn for irrigation purposes, and for the design and sizing of constructed wetlands for wastewater treatment.

The objectives of this study are to:

1. review the methods used for calculating ET from wetlands;
2. determine the data needed to accurately predict ET using an energy balance approach;
3. calculate crop coefficients for bulrush (*Scirpus acutus* Muhl.), cattails (*Thypha latifolia* L.), and floating grass mats, which were composed of Western mannagrass (*Glyceria occidentallis* (Piper) J.C. Nels.) and water foxtail (*Alopercus geniculatus* L.) ; and
4. predict the daily ET from the Oregon State University Dairy Wetland Treatment System (OSUDWTS).

2.3 Previous Wetland Evapotranspiration Studies

Most studies of ET from wetlands have tried to correlate ET data with ecosystem and meteorological variables and then compared the results to open water evaporation (Kadlec and Knight, 1996). Studies have found that wetlands can evaporate more or less than open water. These studies have generally used equations developed for terrestrial systems. Thornwaite's equation for potential ET has been applied to several wetlands with marginal results (Rykiel, 1977; Rykiel, 1984; Kadlec et al., 1987). Others have used the pan evaporation method, which correlates Class A pan evaporation from National Oceanic and Atmospheric Administration (NOAA) climatic centers to wetland ET (Christensen and Low, 1970; Kadlec et al., 1987; Kadlec, J.A., 1986). Kadlec et al. (1987) reported that the Christensen approach (1968) adequately described ET for wetlands in Michigan and Nevada. While these empirical approaches are convenient for predicting wetland ET, they often have a large error associated with them. Dolan et al. (1984) measured wetland ET by continuously monitoring water table elevation and using the diurnal changes to calculate the daily ET. This method, while accurate, does not allow for prediction of future ET.

An accurate approach for predicting ET is to use an energy balance or budget (Bedient and Huber, 1992). The most common equations for calculating ET are modification of the Penman (1948) equation, which is actually a combination of Dalton's Law and an energy balance.

Several studies have evaluated the use of the Penman equation for predicting wetland ET (Allen et al., 1992; Faulkner and Lambert, 1991; Koch and Rawlik, 1993; Lafleur, 1990). In most of these studies a crop coefficient (k_c; the ratio of wetland ET/reference ET) was developed for a specific species of hydrophyte. Crop coefficients vary widely and are dependent on site specifics, such as the size and distribution of the wetland vegetation. Narrow bands of wetland vegetation along streams and lakes or in isolated stands have higher ET rates than wetland vegetation in expansive monotypic stands. This is due to the wind effect ("clothes line effect") which carries away moist air from the plants and increases ET. This wind effect causes the advective losses (the horizontal flux of sensible heat) of energy to become great and increases ET (Anderson and Idso, 1987). Anderson and Idso (1987) also showed that once the surface area covered by vegetation and canopy increased to a particular size the ET rate decreased. They speculate that the larger surface area of vegetation decreased the atmospheric turbulence and decreased advective losses. One must be careful that crop coefficients be used only with the equation for which they were developed.

ET data based on the Penman approach are available in most regions from NOAA climate centers or the U.S. Bureau of Reclamation's AgriMet Stations. The reported ET from these sites is for a reference crop, usually alfalfa or grass, and must be modified by a crop coefficient. As mentioned above, the crop coefficient (k_c) is based on site specifics and also the growing season of the vegetation. Table 2.1 is a list of reported crop coefficients for wetlands, the site specifics, and the ET equation they were used with.

2.4 Methods

The Penman-Monteith equation was used to predict the daily alfalfa reference ET at the OSUDWTS. Meteorological data used in the equation were collected by the U.S. Bureau of Reclamation's AgriMet weather station located at the Oregon State University Hyslop Experiment Station, Corvallis, OR. Daily ET values, calculated using hourly and mean daily meteorological data were compared to see how much error resulted from using mean daily data. These data along with field measurements of ET were used to determine crop coefficients for cattails, bulrush, and floating grass mats.

2.4.1 Study Site

The OSUDWTS is located in Corvallis, Oregon and was designed to treat diluted dairy flushwater. The site consists of six parallel wetland cells 28.1 m x 5.9 m x 0.30 m (Cells 4-9) and a 29.6 m x 10.7 m x 1 m storage pond (Cell 10) (Fig. 2.1). The wetland system was constructed and planted in 1992, began receiving wastewater in October of 1993, and continues to receive wastewater. Treated water is pumped twice daily from pond 10 to a mixing tank where concentrated dairy wastewater is added. The time of day and duration of the pumping of "recycled water" are controlled by a mechanical timer. The concentrated wastewater is loaded from the dairy's pressurized liquid waste handling system using an electric ball valve and electronic timer. The entire volume of the "mixed" wastewater is then loaded to the cells over a period of approximately four hours. The outflows from the wetland ponds drain back into pond 10. The cells are vegetated by a mix of cattails, bulrush, and floating grass (Western mannagrass and water foxtail).

Table 2.1. Crop coefficients (k_c) for various hydrophytes and evapotranspiration methods.

Plant/Wetland Type	Location	Crop Coefficient (k_c)	ET Method	Notes	Source
Typha spp.	Utah U.S.A.	1.6	Penman-Monteith	for isolated stands	Allen et al. 1992
Scirpus spp.	Utah U.S.A.	1.8	Penman-Monteith	for isolated stands	Allen et al. 1992
Freshwater Marsh	Florida U.S.A.	1.00 (0.61-2.5)*	Thornwaite	monthly values reported	Dolan et al. 1984
Freshwater Marsh	Florida U.S.A.	0.79 (0.51-1.06)*	Linacre	monthly values reported	Dolan et al. 1984
Freshwater Marsh	Florida U.S.A.	0.67 (0.34-1.16)*	Pan Evaporation	monthly values reported	Dolan et al. 1984
Dambo Marsh	Africa	0.5	Penman FAO-24	grazed grasses & sedges	Faulkner and Lambert, 1991
Sedges (*Carex* spp.)	Ontario, Canada	0.9	Penman Open Water	subartic coastal wetlands	Lafleur, 1990
Taro Field	Florida U.S.A.	0.74-0.95	Pan Evaporation		Shih and Synder, 1984

* values in parentheses indicate range of monthly coefficients

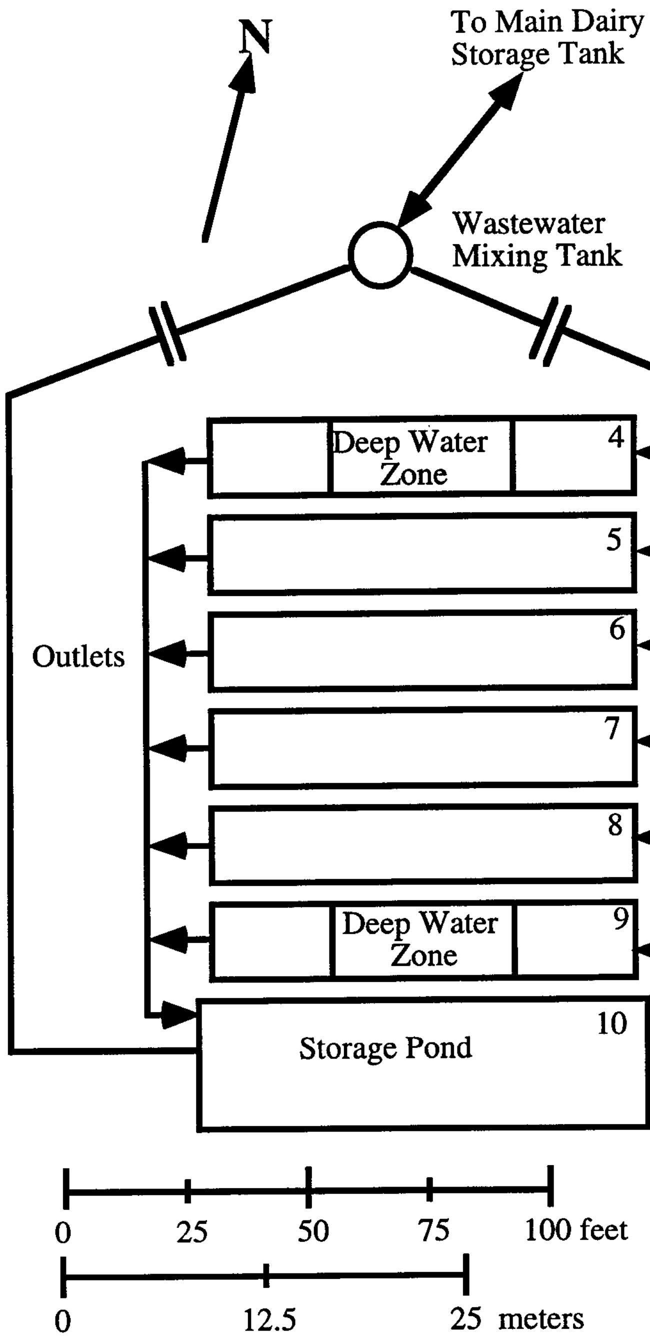

Figure 2.1. Site map of the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Cells 4-9 are the wetland treatment cells and cell 10 is a storage pond. Lines and arrows indicate pipes and the flow path of wastewater.

2.4.2 Calibration Data

In July of 1996, water depths in the wetland cells were measured every morning and night for ten days, when no wastewater or precipitation was being added. This provided a data set to validate water loss due to ET and infiltration. Infiltration rates through the compacted clay liners were measured using a falling head permeameter. Subtracting the daily infiltration from the daily water loss resulted in the daily ET. In addition, the water level in pond 10 was continuously monitored using a Stevens Type F water level recorder for 18 months of the study period. This provided an additional data set to calibrate ET throughout the year. Vegetation cover and height were measured in each wetland cell in July 1996.

2.5 Governing Equations

The fundamental equation for predicting evaporation is Dalton's law. Dalton's Law states that the rate of evaporation is proportional to the difference in vapor pressures at the water surface and the vapor pressure of the surrounding air:

$$E = (e_w - e_z) \cdot (a + b \cdot u) \qquad (2\text{-}1)$$

where, E = evaporation,

e_w = vapor pressure at the water surface,

e_z = vapor pressure at some fixed level above the water surface,

u = wind speed, and

a, b = empirical constants.

The previous equation can also be used for soil-atmosphere and plant-atmosphere interfaces. Meteorological conditions have significant effects on the previous equation and it is necessary to take these conditions into account.

A more accurate approach for predicting ET for terrestrial systems is to use an energy balance or budget (Bedient and Huber, 1992). The data collection for the energy balance approach is more involved, the equations are more complex, and the results are not necessarily more accurate for wetlands than the empirical approaches (Kadlec and Knight, 1996). However, the method uses physically based equations and is less dependent on empirical constants. Use of an energy balance approach also allows for prediction of snow melt and water temperatures in a wetland. The ability to predict temperature is extremely useful because many of the processes in wetlands are temperature dependent. Figure 2.2 shows the energy balance for an element of a wetland and the mathematical summary follows (Kadlec and Knight, 1996):

$$R_n = \rho_w \cdot \lambda_w \cdot ET + H_a + G + (U_o - U_i) + \Delta S \qquad (2\text{-}2)$$

where, $R_{n.}$ = net radiation reaching the ground (MJ/m^2-d),

ρ_w = density of water (kg/m^3),

λ_w = latent heat of vaporization of water (MJ/kg) = 2.453 MJ/kg at 20°C,

ET = water lost to ET (m/d),

H_a = convective transfer to air (MJ/m^2-d),

G = conductive transfer to the ground (MJ/m^2-d),

U_i = energy entering with water (MJ/m^2-d),

U_o = energy leaving with water (MJ/m^2-d), and

ΔS = change of storage of energy with in wetland element (MJ/m^2-d).

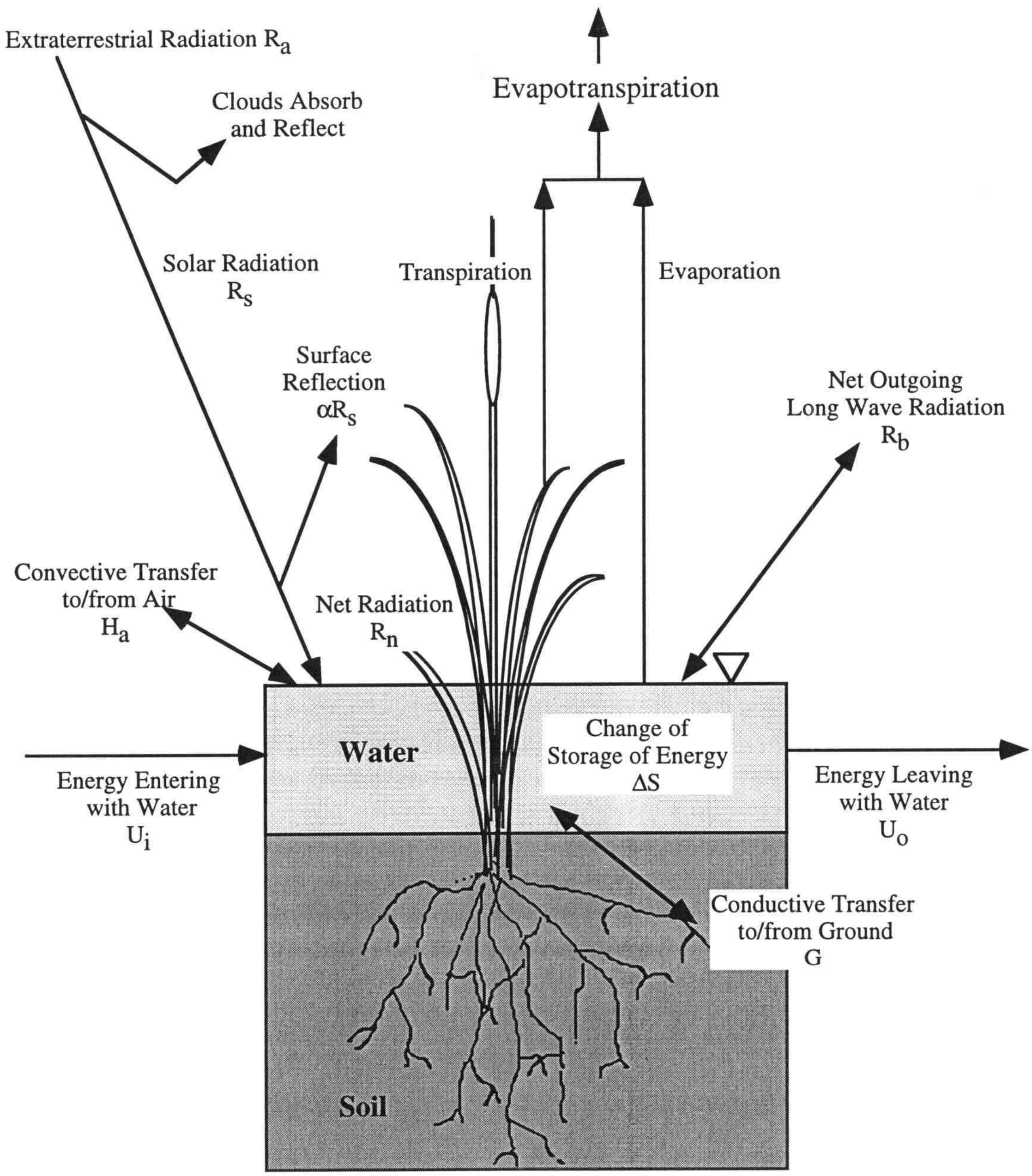

Figure 2.2 Components of a wetland energy balance (based on Kadlec and Knight, 1996).

2.5.1 Penman Equation

The most common ET equation, which uses an energy balance approach, is the Penman (1948) equation, which is actually a combination of Dalton's Law and an energy balance. Several forms of the Penman's equation exist and the Penman-Monteith will be discussed in detail here. This equation differs from other Penman equations in that it includes both aerodynamic and surface resistance terms (ASCE, 1990). This added complexity requires additional crop specific data that are not available for most hydrophytes. However, this equation has been proven to be the most accurate for terrestrial crops and preliminary studies indicate the same will hold true for wetland vegetation (ASCE, 1990; Allen et al. 1992). After manipulation of the energy balance equation, the Penman-Monteith equation states (ASCE, 1990):

$$\lambda E = \frac{\Delta(R_n - G) + \rho \cdot c_p \cdot \frac{(e_a - e_d)}{r_a}}{\Delta + \gamma \cdot \left(1 + \frac{r_c}{r_a}\right)} \qquad (2\text{-}3)$$

where, λE = latent heat flux density (MJ/m^2-d),

R_n = net radiation reaching the ground (MJ/m^2-d),

G = conductive transfer to the ground (MJ/m^2-d),

ρ = air density (kg/m^3),

c_p = specific heat of dry air (J/kg-°C),

e_a = saturation vapor pressure at T_a (kPa),

e_d = saturation water vapor pressure at T_d dew point (kPa),

T_a = air temperature (°C),

T_d = dew point temperature (°C),

r_a = aerodynamic resistance to turbulent transfer of sensible heat and vapor from the plant surface into the atmosphere at the wind measurement height (s/m),

Δ = slope of the saturation vapor pressure curve = $\frac{de}{dT} \cong \frac{e_a - e_d}{(T_a - T_d)}$ (kPa/°C),

γ = the psychometric constant (kPa/°C), and

r_c = bulk canopy resistance (s/m).

For neutral atmospheric stability the previous equation can be manipulated and simplified resulting in (Allen et al., 1992):

$$\lambda ET = \frac{\Delta}{\Delta + \gamma \cdot (1 + r_c / r_a)} \cdot (R_n - G) + \frac{\gamma}{\Delta + \gamma \cdot (1 + r_c / r_a)} \cdot K_1 \cdot \frac{0.622 \cdot \lambda \cdot \rho}{P} \cdot \frac{1}{r_a} \cdot (e_z^0 - e_z) \quad (2\text{-}4)$$

where, λ_w = latent heat of vaporization of water (MJ/kg) = 2.453 MJ/kg at 20°C,

ET = water lost to ET (mm/d),

$$r_a = = \frac{\ln\left[\frac{z_w - d}{z_{om}}\right] \cdot \ln\left[\frac{z_p - d}{z_{ov}}\right]}{(0.41)^2 \cdot u_z}, \quad (2\text{-}5)$$

z_w = the height of the wind speed measurement (m),

z_p = the height of the humidity and temperature measurements (m),

d = zero plane displacement height of vegetation (m),

u_z = wind speed at height z (m/s),

K_1 = dimensionless coefficient = 8.64 x 10^4 for u_z in m/s,

P = atmospheric pressure (kPa),

e_z^0 = saturation vapor pressure of air at height z, and

e_z = water vapor pressure at height z.

The bulk canopy resistance (r_c) for dense uniform crops can be calculated by multiplying individual leaf stomatal resistance (per area of leaf) by the total projected leaf area (Allen et al., 1989):

$$r_c = r_l/(0.5 \cdot \mathrm{LAI}) \qquad (2\text{-}6)$$

where, r_l = the stomatal resistance of a single leaf/unit projected LAI (100m/s for alfalfa and grass), and

LAI = projected leaf area index (defined as the total area of one side of flat leaves (m^2) per m^2 of ground surface).

The 0.5 is based on the observation that only the upper half of a leaf is involved in latent heat exchange in dense uniform stands (Allen et al., 1992). This value, however, does not hold true for vegetation that is not found in dense stands because of the increased mixing and scalar transfer of heat and vapor due to wind moving through the stand. In narrow strips of *Typha* spp. and *Scirpus* spp., Allen et al. (1992) recommend using:

$$r_c = r_l/\mathrm{LAI} \qquad (2\text{-}7)$$

because the entire leaf surface area is actively involved in latent heat exchange.

The height of vegetation changes through the growing season which in turn effects r_c. It is therefore necessary to take in to account the height of the crop. Allen et al. (1989) developed an empirical equation to predict LAI based on the height of alfalfa:

$$\mathrm{LAI} = 1.5 \cdot \ln(h) - 1.4 \qquad (2\text{-}8)$$

where, h = height of alfalfa (must be greater than 5 cm) (cm).

Specific measurements of LAI versus height for hydrophytes do not exist but preliminary studies indicate that the stomatal resistances of a single leaf of *Scirpus* spp., *Thypha* spp., and *Carex* spp. appear to be similar to grass and alfalfa (Allen et al., 1992, Korner et al. 1979). Until further investigations are carried out, it is recommended a value of 100 s/m be used for r_c (Allen et al., 1992).

A second approach is to calculate ET for a reference crop, for which all of the Monteith coefficients are known. Alfalfa and grass are the two most common reference crops. The reference ET can then be multiplied by a crop coefficient (k_c) to give the ET for the vegetation type. The crop coefficient is simply the ratio of crop ET/reference ET. Allen et al. (1992) showed that for *Scirpus spp.* and *Thypha spp.* use of a crop coefficient resulted in better predictions of ET then use of regressed values for stomatal resistance and canopy resistance.

The Penman-Monteith equation is most accurate when hourly meteorological data are used and it has been shown that in some cases significant errors can result from using daily averages (ASCE, 1990). The errors result largely from diurnal changes of wind speed, temperature, and net radiation. Daily averages can give very reliable results but one must be careful to recognize the potential for error. If daily averages are used then (ASCE, 1990):

$$K_1 \cdot \frac{0.622 \cdot \lambda \cdot \rho}{P} = 1710\text{-}6.85 \cdot T \qquad (2\text{-}9)$$

where, T = air temperature (°C).

2.5.1.1 Saturation Vapor Pressure

The Smithsonian Meteorological Tables (List, 1963, List, 1984) allow for saturation vapor pressure (e^{o}) of water to be determined using wet and dry bulb temperatures. A more convenient way of determining e^{o} is to use the following equation (Kadlec and Knight, 1996, modified from Tetens (1930) and Murray (1967)):

$$e^{\circ} = 19.0971 - \frac{5349.93}{T + 273.16} \qquad (2\text{-}10)$$

where, e^{0} = saturation vapor pressure (kPa), and

T = temperature (°C).

This equation fits the Smithsonian Meteorological Tables quite well ($R^2 = 0.99999$) (Kadlec and Knight, 1996).

2.5.1.2 Latent Heat of Vaporization

Latent heat of vaporization of water (λ) changes with temperature but is independent of atmospheric pressure. ASCE (1990) recommend using the following equation to predict λ:

$$\lambda = 2.501 - (2.361 \cdot 10^{-3}) \cdot T \qquad (2\text{-}11)$$

where, λ = latent heat of vaporization (MJ/kg), and

T = temperature (°C).

2.5.1.3 Atmospheric Pressure and Density

In the middle latitudes atmospheric pressure (P) and density (ρ) can be adequately estimated using the following equations (List, 1984):

$$P = 101.3 - 0.01055 \cdot EL, \qquad (2\text{-}12)$$

$$\rho = 1.23 - 0.000112 \cdot EL, \qquad (2\text{-}13)$$

where, P = atmospheric pressure (kPa),

EL = elevation (m), and

ρ = atmospheric density (kg/m^3).

2.5.1.4 Psychometric Constant

ASCE (1990) states that the "psychometric constant (γ) represents a balance between the sensible heat gained from air flowing past a wet bulb thermometer and the sensible heat transformed into latent heat ". The constant can be calculated as (Brunt, 1952):

$$\gamma = \frac{c_p \cdot P}{0.622 \cdot \lambda} \qquad (2\text{-}14)$$

where, γ = psychometric constant (kPa/°C),

c_p = the specific heat of moist air at constant pressure = 0.001013 MJ/kg-°C,

P = atmospheric pressure (kPa),

λ = latent heat of vaporization (MJ/kg), and

0.622 = molecular weight ratio of water/air = 18/29.

2.5.1.5 Net Radiation

In order to use any of the combination equations, net solar radiation (R_n) must be determined. Net solar radiation is rarely directly measured but a number of methods exist for calculating it. All of the methods are based on how much of the extraterrestrial radiation (R_a), which is a function of latitude and season, is lost to absorption and reflection. These equations use either R_a and percent sunshine or solar radiation (R_s) and cloudless-day solar radiation (R_{so}) to calculate R_n. Methods for calculating both R_{so} and R_s are reviewed in ASCE (1990). Solar radiation (R_s) is the amount of radiation that reaches the Earth's surface, which is a function of cloud cover (ASCE, 1990). Various equations exist for calculating R_s as a function of cloud cover and Doorenbos and Pruitt (1977) recommended a generalized equation:

$$R_s = \left(0.25 + 0.5 \cdot \frac{S}{100}\right) \cdot R_a \qquad (2\text{-}15)$$

where, R_s = solar radiation (MJ/m^2-d),

R_a = extraterrestrial radiation (MJ/m^2-d), and

S = percent sunshine.

However, R_s is often measured at weather stations instead of percent sunshine and a different set of equations must be used to calculate R_n. These equations require the calculation or determination of cloudless-day solar radiation. The method that will be described here was developed from calibrations of NOAA climate data in the western USA (Heermann et al. , 1985):

$$R_{so} = A' + B' \cdot \cos[(2' \cdot \pi \cdot d)/365 - C'] \qquad (2\text{-}16)$$

where, R_{so} = cloudless-day solar radiation (MJ/m^2-d),

$A' = 31.54 - 0.273 \cdot Lat + 0.00078 \cdot E$,

$B' = -0.30 + 0.268 \cdot Lat + 0.00041 \cdot E$,

C' = the phase constant for the longest day (170, $\therefore$ $C' = 2.93$) (ASCE, 1990),

Lat = latitude N(°), and

E = elevation (m).

Not only is R_a absorbed and reflected but the Earth also loses longwave radiation (heat) back to the atmosphere. The net outgoing longwave radiation (R_b) is a function of cloud cover, absolute temperature, and moisture content and can be calculated using the general equation (Doorenbos and Pruitt, 1977).

$$R_b = [0.1 + 0.9 \cdot n/N] \cdot \left(0.34 + 1.39 \cdot \sqrt{e_d}\right) \cdot \sigma \cdot (T + 273)^4 \qquad (2\text{-}17)$$

where, R_b = net outgoing long wave radiation (MJ/m^2-d),

n = actual sunshine hours or $R_{s,}$

N = maximum possible sunshine hours or R_{so},

σ = Stefan-Boltzmann constant (4.903 x 10-9 (MJ/ m^2-d-K^4)),and

T = air temperature (°C).

Once R_s and R_b are determined than R_n can be calculated:

$$R_n = (1-\alpha) \cdot R_a - R_b \qquad (2\text{-}18)$$

where, α = short wave reflectance or albedo, and

R_a = extraterrestrial radiation (MJ/m^2-d).

A fraction of the solar radiation is reflected by the surface it strikes, which is termed the albedo (α) of the surface. The albedo of various surfaces and crops have been measured, however, there are little data on the albedo of wetland vegetation. Kadlec and Knight (1996) suggest using the common albedo of green crops (0.23) for wetlands. Prueger (1991), however, predicted that an albedo of 0.17 was most appropriate for cattails. Until further data are available, a value between these two should be used and calibrated to the specific site.

2.5.1.6 Soil Heat Flux

The heat storage and release (G) from soils are minimal on a day to day basis because heat stored during the day is lost to cooling at night. In addition, G is fairly small in relation to R_n and it is sometimes ignored when calculating daily ET (Kadlec and Knight, 1996; ASCE, 1990). If G is used, it can be calculated using the profile of soil temperatures or can be estimated using the equation (ASCE, 1990):

$$G = -\rho \cdot c \cdot d \cdot \frac{(T_i - T_{i+1})}{\Delta t} \qquad (2\text{-}19)$$

where, G = conductive transfer to ground (MJ/m^2-d),

ρ = density of soil (Mg/m^3),

c = specific heat of soil (kJ/kg-°C),

d = depth of temperature exchange (m),

T_i = equals the mean air temperature for time period i (°C),

T_{i+1} = equals the mean air temperature for time period i +1 (°C), and

Δt = time in days between the midpoints of the two periods.

The time periods i and i+1 should be at least 10 days long and a value of 4.2 is often used for $\rho \cdot c \cdot d$ (ASCE, 1990). Kadlec and Knight (1996) estimated G for wetland soils to be on the order of 0.5 MJ/m^2-d while R_n at the peak of summer is on the order of 10 to 20 MJ/m^2-d.

2.5.2 Evapotranspiration Calculation

Once all of the terms required for the Penman-Monteith equation have been measured or calculated, ET can be directly determined using the following equation:

$$\lambda ET = \frac{\Delta}{\Delta + \gamma \cdot (1 + r_c / r_a)} \cdot (R_n - G) + \frac{\gamma}{\Delta + \gamma \cdot (1 + r_c / r_a)} \cdot K_1 \cdot \frac{0.622 \cdot \lambda \cdot \rho}{P} \cdot \frac{1}{r_a} \cdot (e_z^0 - e_z). \tag{2-20}$$

Computer models or use of a spreadsheet greatly simplifies the calculation process and allows for various estimates of coefficients to be evaluated.

2.6 Results

Daily reference ET was calculated for the month of July 1996 using both hourly and mean daily meteorological data (Fig. 2.3). It was found using a paired t-test that these two methods were statistically different (p-value < 0.0001). The ET calculated using mean daily data predicting 0.6 mm/d less ET than when hourly data was used. It was decided that this error was small enough that daily mean meteorological data could be used for ET

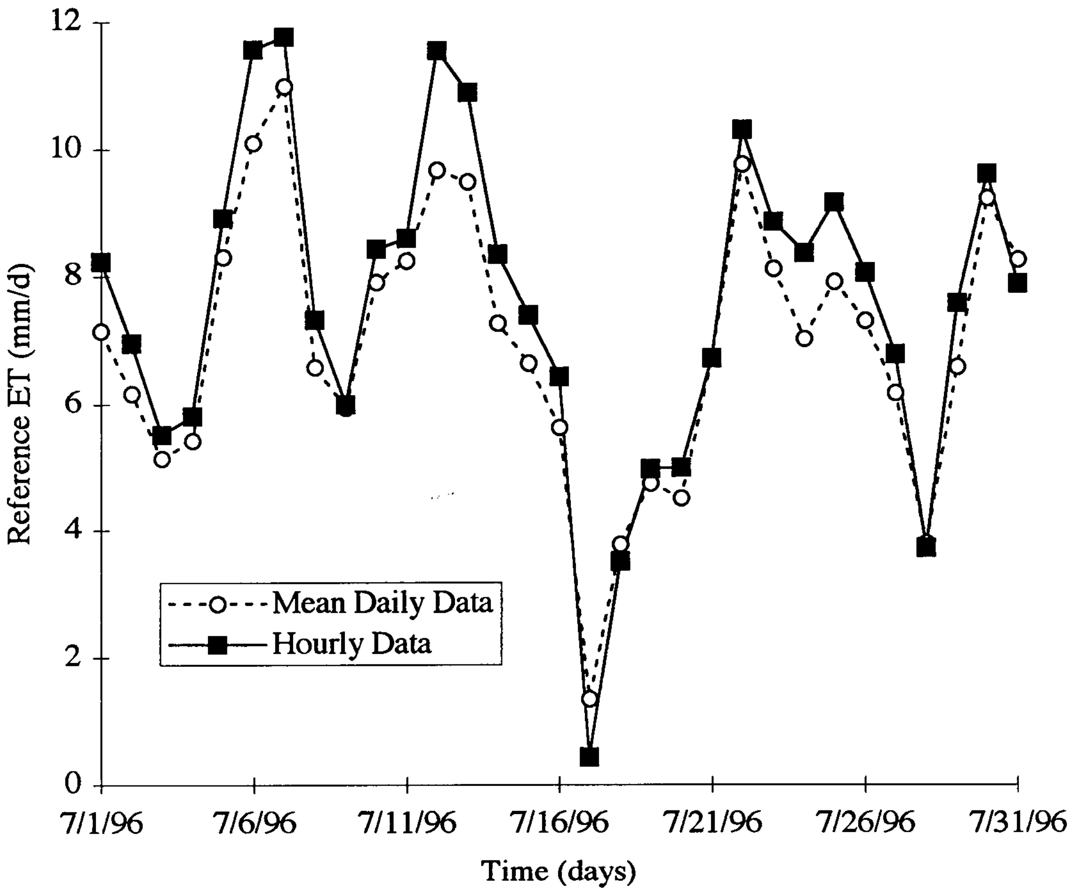

Figure 2.3 Calculated alfalfa reference evapotranspiration at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

calculations. Monthly meteorological data and predicted reference ET for 1996 is summarized in Table 2.2.

The average infiltration rate for the wetland cells was 0.96 mm/d. The measured ET for each of the wetland cells during July 4-14, 1996 is shown in Table 2.3. Vegetation type and cover varied for each of the wetland cells (Table 2.4). A crop coefficient of 0.8 was selected for the open water areas (ASCE, 1990). A linear regression, minimizing the sum squared error was used to fit crop coefficients (k_c) for each vegetation type (Table 2.4). The resulting k_c values were 0.57, 1.72, and 2.32 for the floating grass mats, bulrush, and cattails, respectively. The average ET coefficient for each wetland cell was then calculated using the percent cover and corresponding k_c for each vegetation type. For example, wetland cell 4 had a k_c of (Table 2.4):

$$k_4 = (\%\text{Grass})\ k_g + (\%\text{Cattails})\ k_c + (\%\text{Bulrush})\ k_b + (\%\text{Open Water})\ k_o$$

$$k_4 = (0.25) \cdot 0.57 + (0.12) \cdot 1.72 + (0.48) \cdot 0.2.32 + (0.15) \cdot 0.8$$

$$k_4 = 1.58$$

where, k_4 = average crop coefficient for pond 4,

k_g = crop coefficient for grass,

k_c = average crop coefficient for cattails,

k_b = average crop coefficient for bulrush, and

k_o = average crop coefficient for open water.

Use of these coefficients resulted in a fairly good fit of ET for all ponds ($R^2 = 0.55$) (Fig. 2.4). These crop coefficients are for the maximum growth stage of the vegetation. For annual ET predictions, k_c must be adjusted to the growth stage of the vegetation. The k_c

Table 2.2. 1996 average monthly meteorological data and monthly precipitation and reference evapotranspiration (ET) for Corvallis, OR. Data was collected by Bureau of Reclamation Agrimet weather station at Hyslop Experiment Station, Corvallis, OR. The reported evapotranspiration was predicted for alfalfa using the Penman-Monteith equation.

Month	Mean Daily Temp. (°C)	Max. Daily Temp. (°C)	Min. Daily Temp. (°C)	Daily Solar Radiation (MJ/m^2)	Daily Mean Dew Point Temp. (°C)	Daily Wind Run (km)	Total Precipitation (cm)	Total Predicted Reference ET (cm)
Jan	5.5	8.1	3.3	3.5	4.6	303.5	26.3	1.8
Feb	5.8	10.1	2.4	7.3	2.9	329.7	35.8	5.0
Mar	8.5	13.2	4.1	11.2	5.7	254.9	9.6	6.8
Apr	10.7	16.2	6.1	14.9	8.0	366.9	12.4	8.7
May	11.3	16.7	6.4	19.3	8.2	149.2	11.1	10.1
Jun	15.3	22.3	8.6	25.6	10.4	168.8	2.4	15.1
Jul	20.5	29.2	12.2	26.8	13.3	189.5	2.6	21.5
Aug	19.1	27.7	11.2	23.6	12.1	175.1	0.4	19.1
Sep	14.9	21.9	8.7	16.6	10.2	151.6	6.4	11.6
Oct	12.3	17.9	7.3	9.6	10.3	137.1	8.4	5.0
Nov	7.1	10.7	4.4	4.4	6.7	122.6	25.9	1.6
Dec	5.9	8.4	3.2	2.7	5.1	197.4	43.5	1.2
Mean	11.4	16.9	6.5	13.8	8.1	212.2		
Total							184.8	107.4

Table 2.3. Evapotranspiration (mm/d) for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR, July 1996.

	Evapotranspiration (mm/d)						
Date	Cell 4	Cell 5	Cell 6	Cell 7	Cell 8	Cell 9	Cell 10
7/4/96	11.7	13.3	18.1	8.6	5.4	8.6	2.2
7/5/96	11.7	7.0	11.7	8.6	8.6	8.6	5.4
7/6/96	21.3	18.1	21.3	8.6	11.7	8.6	8.6
7/9/96	13.7	14.1	16.5	10.2	11.7	9.4	---
7/10/96	13.7	14.1	16.5	10.2	11.7	9.4	---
7/11/96	17.3	10.2	18.1	7.0	11.7	3.8	---
7/14/96	11.7	11.7	19.7	11.7	16.5	7.0	---

Table 2.4. Vegetation composition and fit of crop coefficients for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR, July 1996.

	Percent Coverage by Vegetation Type (%)				Average k_c
Wetland Cell	Grass	Bulrush	Cattails	Open Water	for each cell
4	25	12	48	15	1.58
5	15	79	6	0	1.59
6	2	0	98	0	2.28
7	35	5	30	30	1.22
8	20	20	60	0	1.85
9	30	40	0	30	1.10
10	0	0	0	100	0.80
k_c for Veg. Type =	0.57	1.72	2.32	0.80	
				Average k_c =	1.60

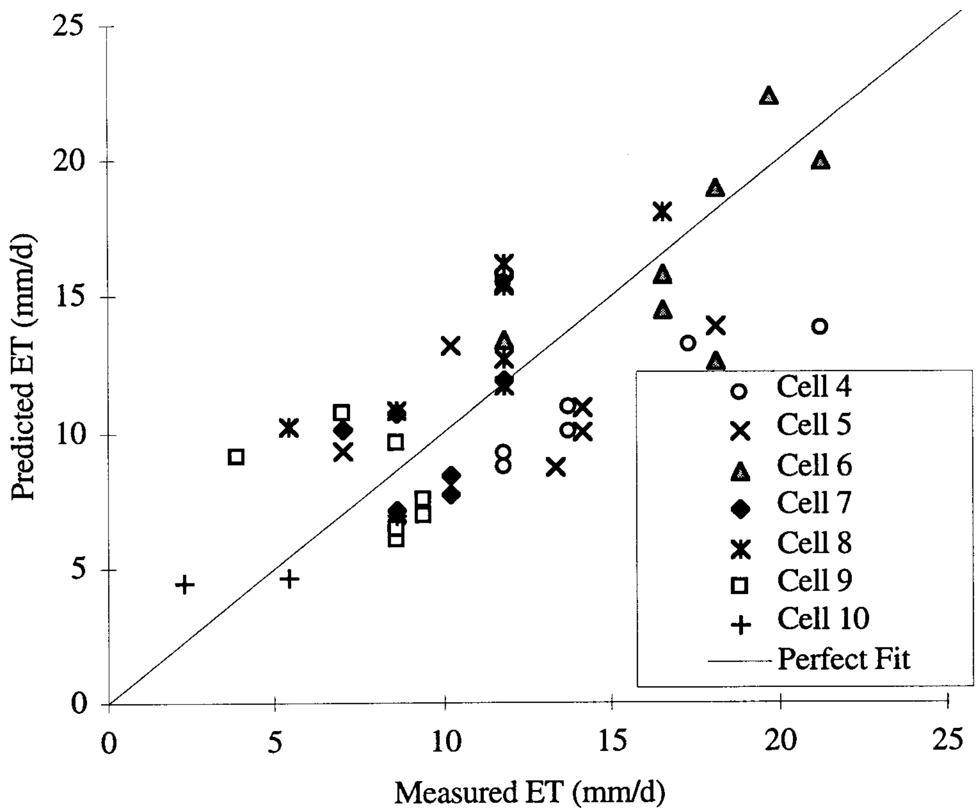

Fig 2.4. Scatterplot of measured versus predicted evapotranspiration for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR, July 1996 ($R^2 = 0.55$).

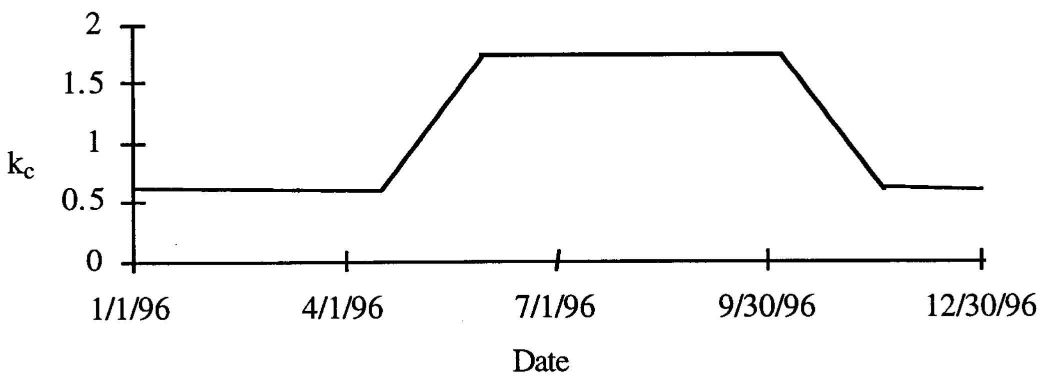

Figure 2.5. The k_c curve for *Typha latifolia* at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

curves were developed for each vegetation type using the date of last killing frost, peak growth, senescence, and first killing frost (Fig. 2.5). A k_c value of 0.6 was used for the winter months and was derived through calibration of the water budget for the site (see Chapter 3). The ET rate throughout the year was validated using an overall water budget model (Fig 2.6) (see Chapter 3). An excellent fit was achieved and provides evidence that the ET predictions were accurate ($R^2 = 0.95$) (Fig. 2.7).

2.7 Discussion

Use of daily mean meteorological data did not result in large errors of predicted ET compared to using hourly data to predict ET.

The k_c for the floating grass mats was low (0.57). The floating grass mats occurred in dense uniform clumps and were an average of 10 cm high. It is hypothesized that the low k_c was a result of two factors:

1. because the mats are short, they are protected from the wind by both the wetland banks and surrounding vegetation, which causes the advective loss to be low, and
2. the grass in many of the ponds is growing intertwined with cattails and bulrush, which shade the grass from direct solar radiation, therefore transpiration is lower.

The combination of these two factors may explain the low k_c for the floating grass mats.

The bulrush were 1.5 to 2 meters high and the k_c for bulrush was 1.72, which is very close to the k_c of 1.8 reported by Allen et al. (1992) for 1.5 m tall bulrush. This high value is most likely caused by extremely high advective losses, which are typical of tall isolated stands of vegetation (Allen et al., 1992).

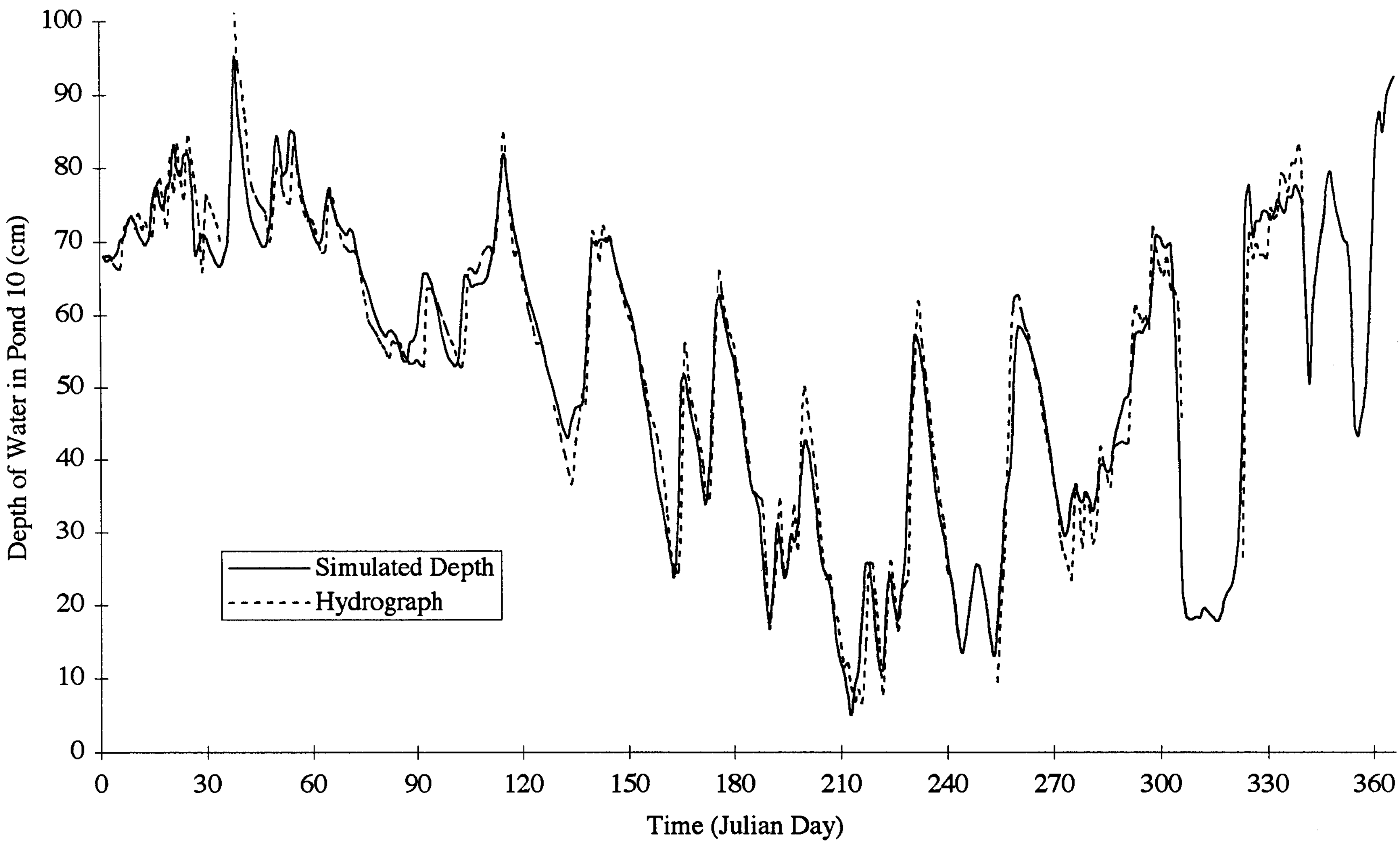

Figure 2.6. 1996 simulated versus actual water depth in Cell 10 at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

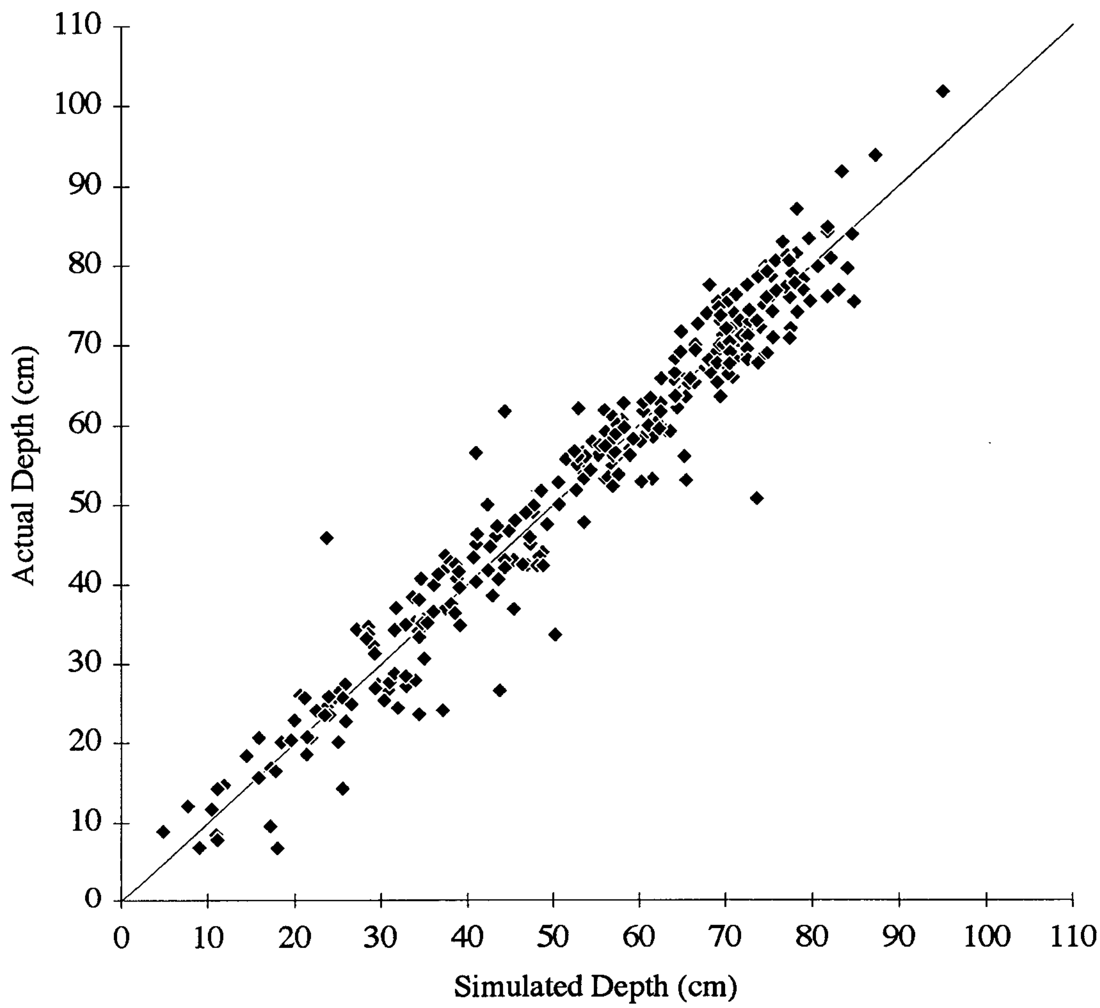

Figure 2.7. Scatterplot of daily actual depth versus simulated depth for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR ($R^2 = 0.95$).

The k_c for cattails was 2.32, which is an extremely high value. This high of a value is questionable and simply may be an artifact of the regression procedure used and the small sample size. However, Allen et al. (1992) reported a k_c of 1.6 for 1.5 m tall cattails in isolated stands and the cattails at the OSUDWTS were 0.5 to 1.5 m taller than the cattails in Allen et al. (1992) study. This additional height increases the surface area exposed to wind and solar radiation and may explain the extremely high value of k_c.

The winter k_c value of 0.6 is lower than open water ($k_c = 0.8$) and it is hypothesized that the fallen vegetation protects the water from both solar radiation and wind, which lowers the evaporative losses.

2.8 Conclusions

Use of crop coefficients and the Penman-Monteith alfalfa reference ET provide an accurate method for predicting wetland ET. Tall isolated stands of wetland vegetation have extremely high ET rates, caused by high advective losses. Expansive monotypic stands of these vegetation types have lower ET rates, because advective losses are decreased. Short floating mats of wetland vegetation have lower ET rates than open water. The lower ET rate for the floating grass is hypothesized to be because of shading by taller vegetation and low advective losses.

Crop coefficients were found for cattails, bulrush, and floating grass mats. Care should be taken if using these crop coefficients for other sites because the coefficients may be an artifact of the regression procedure and the small sample size used in this study. The average ET for all of the wetland cells was 1.6 times as great as Penman-Monteith reference ET. Wetland ET is largely dependent on vegetation type, density, and distribution. Additional studies are needed to: determine if the crop coefficients found in this study are accurate; develop crop coefficients for other hydrophytes; and measure the ET rates for different wetland types.

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3. A Hydrology Model for Constructed Wetlands

3.1 Abstract

A water budget is critical for calculating a contaminant mass balance in a wetland treatment system. Seasonal variability in rainfall and evapotranspiration have a dramatic effect on the overall water budget and can cause a significant impact to the treatment performance. A simple mathematical model was developed to dynamically predict the daily to annual water budget of a wetland system. The model simulates all major hydrologic pathways and is easily adapted to any wetland.

The model was calibrated with data collected at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. The model accurately predicted the water depth in the system throughout the year ($R^2 = 0.95$). Seasonal rainfall and evapotranspiration caused average monthly detention time to vary from the theoretical detention time (4.36 days) by as much as -18% in winter and 9% in summer.

The model provides a tool for easily predicting a detailed water budget of a constructed wetland, a means for evaluating the effect of wetland design and configuration on the water budget, and a foundation for additional treatment submodels.

Keywords: evapotranspiration; precipitation; infiltration; simulation; detention time, wetland, water budget

3.2 Introduction

Hydrology is the single most important factor that determines the establishment and maintenance of wetlands and wetland processes (Mitsch and Gosselink, 1993). The hydrology of a wetland creates the unique physiochemical conditions that are neither completely terrestrial nor aquatic. The hydrologic conditions determine many of the abiotic conditions in wetlands, such as soil anaerobiosis. The abiotic conditions affect biotic factors such as flora and fauna presence and distribution. The flora and fauna may in turn influence the hydrology.

Sources of inflow to wetlands are streamflow, runoff, groundwater, and precipitation. Losses of water from wetlands occur through streamflow, infiltration, and evapotranspiration. The importance of each of these elements varies from site to site and is dependent on climatological and geological conditions of the area and the position of the wetland in the landscape. Constructed or treatment wetlands have an additional source of water, the wastewater. The processes for both natural and constructed wetlands are the same but generally the wastewater is the primary source of water to a constructed wetland. In addition, constructed wetlands generally have uniform basins with an impermeable liner, which simplifies the hydrologic budget.

A hydrologic budget is a summation of all inflows and outflows and is a good approach for determining the magnitude and importance of each element of the hydrologic cycle. A water budget is critical when determining mass balances of specific wastewater constituents and the resulting treatment efficiency. Many constructed wetlands have been evaluated by comparing inlet and outlet concentrations without taking into account the effect of hydrologic gains or losses. This can result in erroneous conclusions and makes comparison of data from different wetlands almost impossible. The water gains or losses not only have a dilution or concentration effect but they also change the detention time, which affects the opportunity for treatment of the wastewater.

The objective of this study is to develop a model that simulates the relative importance and magnitude of each element in the hydrologic cycle for a wetland. This model was developed and calibrated using field data from a constructed wetland system in Corvallis, OR. Simulation output includes daily summaries for each element of the hydrologic cycle and the detention time for the wetland. Simulations were also carried out to determine the effect of varying wetland area and depth on the overall water budget. This model can be used to evaluate existing wetlands or to predict the water budget of a future wetland. This hydrology model could also serve as the backbone for a biological submodel.

3.3 Previous Hydrology Models

Hydrology models have been developed for all types of wetland systems and vary from the very simple to the very complex (Costanza and Sklar, 1985; Mitsch, 1988). Surface flows, groundwater flows, solar radiation, precipitation, atmospheric moisture, and evapotranspiration are important hydrologic processes that should be included in a hydrology model (Duever, 1988). Each wetland has its own unique hydroperiod, which is its seasonal pattern of water levels. The hydroperiod is the result of the balance of inflows and outflows, the surface topography, subsurface soils, geology, and groundwater conditions (Mitsch and Gosselink, 1993). Inflows consist of surface runoff, groundwater inflows, precipitation, and possibly tidal inflows. Outflows consist of evapotranspiration, groundwater outflows, tidal outflows, and surface outflows. Topography, soils, geology, and groundwater conditions all determine the ability of the wetland to hold water. In addition to the previous parameters, rainfall intensity and duration, along with the size, shape, topography, and infiltration rate of the watershed and wetland determine the hydrograph. Small watersheds tend to have rapidly pulsing hydrographs, while larger watersheds generally have smooth hydrographs (Carter, 1979). The importance of each parameter changes on a site to site basis and also for different wetland types. Therefore, it is often necessary to monitor a wetland and collect data about watershed characteristics, local climate, wetland volume, hydrograph, soils, and groundwater before modeling can occur. While a hydrology model may stand alone and be the only focus of study, it is usually a submodel of another model. Because hydrology is the driving force behind every wetland, it must be understood to model chemical cycles, solids retention, primary productivity, and plant succession.

3.4 Methods

The general approach that was followed for construction of the hydrology model was that described by Jørgensen (1986). Odum diagrams were used to conceptualize the system and helped determine the important parameters and data requirements (Odum, 1983). Once a general diagram was drawn a model was constructed using STELLA™ (Structural Thinking, Experimental Learning Laboratory with Animation) language and software (Richmond et al., 1987). The model was developed and calibrated using hydrologic data collected at a constructed wetland system treating dairy wastewater. Model parameters were taken from published research, measured in the field, or estimated based on the field data.

The model predicts the daily, monthly, and annual water budget. Simulation time for the model is one year with a time step of one day. The simulations used a fourth-order Runge-Kutta technique and an integration interval of 0.1 day. After calibration, the hydrology model was used to simulate field conditions for different years and various scenarios of different water depths and surface areas.

3.4.1 Study Site

The Oregon State University Dairy Wetland Treatment System (OSUDWTS) is located in Corvallis, Oregon and was designed to treat diluted dairy flushwater. The site consists of six parallel wetland cells 28.1 m x 5.9 m x 0.30 m (Ponds 4-9) and a 29.6 m x 10.7 m x 1 m storage pond (Pond 10) (Fig. 3.1). The wetland system was constructed and planted in 1992, began receiving wastewater in October of 1993, and continues to receive wastewater. Treated water is pumped twice daily from pond 10 to a mixing tank where concentrated dairy wastewater is added. The time of day and duration of the pumping of "recycled water" are controlled by a mechanical timer. The concentrated wastewater is loaded from the dairy's pressurized liquid waste handling system using an electric ball

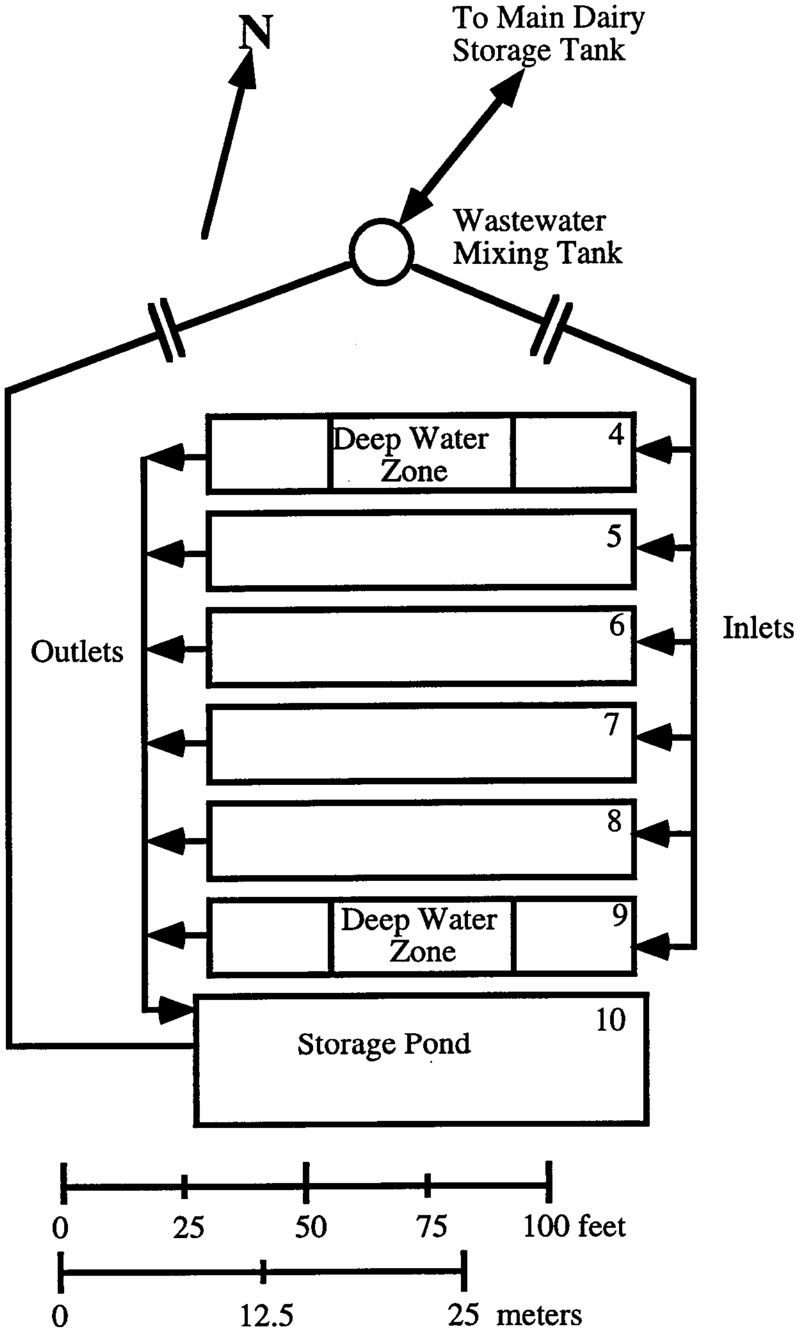

Figure 3.1. Site map of the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Cells 4-9 are the wetland treatment cells and cell 10 is a storage pond. Lines and arrows indicate pipes and the flow path of wastewater.

valve and electronic timer. The entire volume of the "mixed" wastewater is then loaded to the cells over a period of approximately four hours. The outflows from the wetland ponds drain back into pond 10. The cells are vegetated by a mix of bulrush (*Scirpus acutus* Muhl.), cattails (*Thypha latifolia* L.), and floating grass mats, which were composed of Western mannagrass (*Glyceria occidentallis* (Piper) J.C. Nels.) and water foxtail (*Alopercus geniculatus* L.).

3.4.2 Calibration Data

Data collected at OSUDWTS were used to develop and calibrate the model. Daily evapotranspiration was calculated using the Penman-Monteith equation (see Chapter 2). Infiltration rates through the compacted clay liners were measured using a falling head permeameter. Water depths were measured in the ponds for a two week period when no wastewater or precipitation were being added. This provided a data set to validate water loss due to evapotranspiration and infiltration. The depth to volume relationship of the storage pond was determined by filling and emptying the pond at a known flow rate. The area of the catchment, wetland ponds, and storage pond were all measured in the field. Pumping times, duration, and rates were all verified in the field. Water was occasionally pumped out of or into pond 10 to prevent the pond from overflowing or drying out. The volume of water pumped out or into pond 10 was recorded. Water levels in pond 10 were continuously monitored using a Stevens Type F water level recorder for 18 months of the study period. These data were used for final calibration and validation of the hydrology model. Additional information, about the wetland treatment and twelve water quality parameters monitored, is given in Skarda et al. (1994) and can be found in the Livestock Wastewater Treatment Wetland Database (Knight et al., 1996).

3.5 Modeling/ Governing Equations

The hydrology model predicts the water budget of a wetland given the climatic conditions and site specifications of the wetland. The overall water budget is calculated using the following equation:

$$dV_{wet}/dt = Q_s + Q_r + Q_{sm} + Q_w \pm Q_g + P \cdot A_{wet} - Q_b - Q_{out} - ET \cdot A_{wet} \qquad (3.1)$$

where, V_{wet} = wetland water volume (m^3),

Q_s = streamflow (m^3/d),

Q_r = runoff (m^3/d),

Q_{sm} = runoff from snowmelt (m^3/d),

Q_w = wastewater loading rate (m^3/d),

Q_g = groundwater interaction (m^3/d),

P = direct precipitation (m/day),

A_{wet} = surface area of wetland (m^2),

Q_b = bank losses (m^3/d),

Q_{out} = outflow rate (m^3/d), and

ET = evapotranspiration (m/d).

The importance of each of these terms will vary from site to site and season to season but all must be included when making a general model.

3.5.1 Wetland Specifications

In order to calculate the overall wetland water budget and theoretical detention time, it is necessary to know the wetland volume, depth, and area. These effect the amount of direct precipitation, evapotranspiration, infiltration, and runoff. If the wetland basin is not uniform or the depth is not kept constant, it is necessary to know the water volume to depth relationship. It is also important to determine the volume to area relationship because if the surface area of the wetland changes, then the amount of direct precipitation, evapotranspiration, and infiltration change. The depth versus volume and area versus volume relationships are easy to determine in wetlands with simple geometry but can be more difficult to determine in wetlands with non uniform geometry and depths. Site surveys and design plans are a convenient way to calculate the depth versus volume and area versus volume relationships. Another convenient and accurate way to measure these relationships is to pump water into or out of the wetland at a known rate and measure the change in depth and area.

The wetland volume, depth, surface area, and size of the catchment are all inputs into this model. It is also necessary to specify the depth versus volume and area versus volume relationships.

3.5.2 Inputs From Surface

Water enters wetlands from overland flow in three major ways: streamflow, runoff from rainfall and snowmelt, and direct loading of wastewater. Generally, streamflow is not important in constructed wetlands except for nonpoint source and stormwater wetlands. Runoff is dependent on the catchment size of the constructed wetland and is usually a small fraction of the total surface flows. Direct loading of wastewater into a constructed wetland, as a rule, is the major source of surface flow. However, it is important to consider all of the pathways when constructing a complete water budget.

3.5.2.1 Stream Flow

Streamflow does not generally occur in constructed wetlands except for stormwater driven wetlands. If the wetland hydrology is storm driven, it is necessary to measure streamflow. If there is a single streamflow the easiest way to measure flow is to have a weir and stage level recorder. The combination of weir size and stage height allows for accurate calculation and re-creation of storm hydrographs (Chow, 1964). If the channel is irregular and no weir exists, then the cross sectional area, velocity, and depth must be measured to calculate flow rates. Both methods require temporal measurements at different discharges, with greater frequency of measurements giving more accurate results. If enough data are collected, stage discharge curves may be drawn which allows for prediction of flows at different stage heights. If the rainfall is known you can also create a rainfall discharge curve which would allow for prediction of discharge flow based on rainfall.

If multiple stream flows enter the wetland the above method must be used for each incoming streamflow. This model assumes that the researcher has measured any incoming stream flows and the data will be entered as a daily flow (Q_s) in m^3/day.

3.5.2.2 Runoff

Runoff is the water that directly enters the wetland by overland flow. The area that collects and delivers waters to the wetland is called the catchment. Generally, constructed wetlands have a well defined catchment that is determined by berms or access roads. The size of the catchment for small and medium sized wetlands is approximately 25% the size of the wetland surface area (Kadlec and Knight, 1996). Runoff occurs when the rainfall rate exceeds the infiltration rate. The infiltration rate is based on the soil's hydraulic conductivity and the antecedent moisture conditions. The Green-Ampt Equation can be used to predict the time to soil saturation and the rainfall rate needed before runoff occurs. This equation requires that soil moisture and the soil wetting front be modeled. This level of detail is not usually warranted for predicting a wetland water budget.

The rational method is an alternative for predicting runoff. This method predicts runoff based on the area of the catchment times the precipitation rate times a runoff coefficient:

$$Q_r = k_r \cdot P \cdot A_{cat} \qquad (3\text{-}2)$$

where, Q_r = runoff (m^3/d),

k_r = runoff coefficient,

P = direct precipitation (m/day), and

A_{cat} = catchment area (m^2).

Constructed wetland catchments generally consist of nearly impermeable berms and access roads, thus runoff coefficients will be high with 80 to 100% of the rain water being delivered to the wetland (Kadlec and Knight, 1996). This method, however, over predicts runoff during low intensity rain events and under predicts runoff during high intensity or long duration events.

As a compromise to these two methods, runoff will be predicted using two scenarios in this model. If rainfall exceeds a specified rate then the rational method will be used. Otherwise rainfall will infiltrate into the soil until the potential storage capacity of the catchment is exceeded. Once the storage capacity is exceeded then runoff occurs. The potential storage volume of the catchment is calculated as:

$$V_{cat} = A_{cat} \cdot D_{cat} \cdot \eta \qquad (3\text{-}3)$$

where, V_{cat} = potential storage volume of catchment (m^3),

A_{cat} = catchment area (m^2),

D_{cat} = depth to impermeable layer or groundwater (m), and

η = effective porosity of soil (decimal fraction).

Water will be removed from the catchment based on the infiltration rate of the soils and the evapotranspiration rate, which must be specified in the model.

3.5.2.3 Loading Rates

Loading rates to wetlands may be reported in units of depth/day or volume/day. The hydraulic loading rate generally refers to the units of depth/day. The hydraulic loading rate (HLR) is calculated using the following equation:

$$q_w = Qw / A_{wet} \qquad (3\text{-}4)$$

where, q_w = wastewater hydraulic loading rate(cm/d, m/d, cm/yr m/yr),

Q_w = wastewater loading rate (m^3/d, m^3/yr), and

A_{wet} = surface area of wetland (m^2).

This loading rate is most often used to refer to the inlet loading but one must be careful because q may also refer to an interior local volumetric flow rate (Kadlec and Knight, 1996). The advantage of using the hydraulic loading rate versus water loading rate is that the hydraulic loading rate is directly proportional to the size of the treatment system. This allows for comparison of sites with the same q_w, even though their wastewater loading rates (Q) or surface area may be very different.

Wastewater loading may occur continuously or intermittently. Intermittent loading may be daily, weekly, seasonal, or storm event driven. If the intermittent loading occurs over a short period, days up to a week, then q_w refers to the time average flow rate. If loading is seasonal then q_w is used to refer to the loading rate during the time of operation. This model requires the input of wastewater loading rate (Q_w), duration, and frequency.

3.5.2.4. Snowmelt

In cold climates, snowmelt is an additional source of surface water. Snowmelt contributes water to a wetland system when temperatures are above freezing. The rate of snowmelt is dependent on net solar radiation, net longwave radiation exchange, conduction and convection transfer of heat to or from overlying air, condensation of water vapor from the overlying air, conduction from the overlying soil, and heat supply by incident rainfall (Bedient and Huber, 1992). Chow (1964) describes a simplified equation for predicting snowmelt based on the area of the catchment, daily maximum temperature, and average temperature:

$$Q_{sm} = 0.0254 \cdot [0.03 \cdot (T_{mean}-24)+0.02 \cdot (T_{max}-27) \cdot A_{cat} \quad (3\text{-}5)$$

where, Q_{sm} = snowmelt (m^3/d),

T_{mean} = mean daily temperature (°C),

T_{max} = maximum daily temperature (°C), and

A_{cat} = catchment area (m^2).

Snow accumulation and melt are modeled by having a snow storage state variable. Snow is stored in the catchment until snowmelt occurs. Snowmelt occurs until the volume of water in the snow is depleted.

3.5.3 Groundwater Interactions

Wetlands can either gain or lose water to the groundwater depending on the head and infiltration rates in the wetland and aquifer. Constructed wetlands generally have an impermeable membrane or nearly impervious clay liner. Thus, gains or losses to groundwater are generally small. Infiltration can occur through the bottom of the wetland (direct infiltration) or through the banks of the wetland (Fig. 3.2).

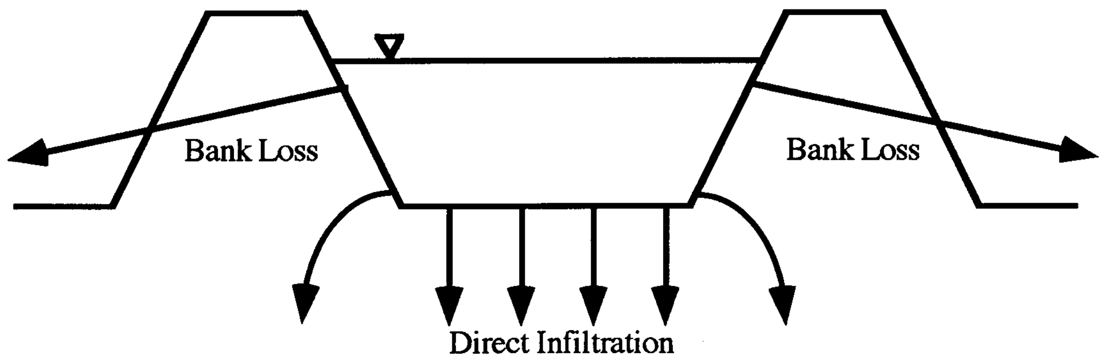

Figure 3.2. Cross section of wetland cell showing water loss due to direct infiltration and bank loss.

3.5.3.1 Bank Losses

Losses through the banks in constructed wetlands should be small if impervious materials are used such as compacted clay. In some cases, such as sandy soils, bank loss may be significant and it is important to account for it. Bank loss can be determined by using the following empirical equation (Burns and McDonnell, 1992):

$$Q_b = k_b \cdot L_b \cdot (H_{wet} - H) \qquad (3\text{-}6)$$

where, Q_b = bank losses (m^3/d),

k_b = empirical loss coefficient (m/d),

L_b = thickness of berm (m),

H_{wet} = wetland water elevation (m), and

H = external water level (m).

In order, to use this equation the empirical loss coefficient must be found by calibrating the measured water loss with head difference inside and outside the bank.

3.5.3.2 Infiltration

The soils under a wetland may range from unsaturated to fully saturated. Most constructed wetlands have an impermeable liner or nearly impervious compacted clay layer under them, which usually results in the underlying soils being unsaturated. In this case infiltration is fairly easy to calculate using Darcy's equation. Darcy's law states that the flow of groundwater is proportional to the hydraulic gradient and the hydraulic conductivity of the soil the water is passing through. Assuming that the underlying soils are unsaturated, the infiltration loss can be calculated as:

$$Q_g = k_g \cdot A_{wet} \cdot (D_{wet}+D_{lin})/(D_{lin}) \qquad (3\text{-}7)$$

where, Q_g = groundwater interaction (m^3/d),

k_g = saturated hydraulic conductivity of liner (m/d),

A_{wet} = surface area of wetland (m^2),

D_{wet} = wetland water depth (m), and

D_{lin} = thickness of liner (m).

If the soils are saturated or the groundwater level is higher than the wetland water level, then infiltration losses and gains are more difficult to calculate. Extensive data on local groundwater levels, soil conductivities by horizon, and local groundwater flows are needed for calculation of the gains or losses. This situation should be rare for constructed wetlands and it will be assumed in this model that the unsaturated case will exist.

3.5.4 Evapotranspiration

Evapotranspiration (ET) is the loss of water to the atmosphere by the process of evaporation and transpiration. Evaporation is the vaporization of water from water and soil surfaces. Transpiration is the passing of water through vascular plants to the atmosphere. Dalton's Law states that the rate of evaporation is proportional to the difference in vapor pressures at the water surface and the vapor pressure of the surrounding air:

$$E = (e_s - e_a) \cdot (a + bu) \qquad (3\text{-}8)$$

where, E = evaporation,

e_s = vapor pressure at the water surface,

e_a = vapor pressure at some fixed level above the water surface,

u = wind speed, and

a, b = empirical constants.

The previous equation can also be used for soil-atmosphere and plant-atmosphere associations. Meteorological conditions have significant effects on the previous equation and it is necessary to take these conditions into account.

Most studies of ET from wetlands have tried to correlate ET data with ecosystem and meteorological variables and have compared the results to open water evaporation (Kadlec and Knight, 1996). Studies have found that wetlands can evaporate more or less than open water. These studies have generally used equations developed for terrestrial systems. Thornwaite's equation for potential ET has been applied to several wetlands with marginal results (Rykiel, 1977; Rykiel, 1984; Kadlec et al., 1987). Others have used the pan evaporation method, which correlates Class A pan evaporation from National Oceanic and Atmospheric Administration (NOAA) climatic centers to wetland ET (Christensen and Low, 1970; Kadlec et al., 1987; Kadlec, J.A., 1986). Kadlec et al. (1988) reported that the Christensen approach (1968) adequately described ET for wetlands in Michigan and

Nevada. While these empirical approaches are convenient to use, they are often inadequate for predicting wetland ET. Dolan et al. (1984) measured wetland ET by continuously monitoring water table elevation and using the diurnal changes to predict the daily ET. This method, however, does not allow for prediction of future ET.

A more accurate approach for predicting ET is to use an energy balance or budget (Bedient and Huber, 1992). The most common equations for calculating ET are modification of the Penman (1948) equation, which is actually a combination of Dalton's Law and an energy balance. For a complete discussion of the Penman equation and its use for predicting ET from wetlands see Chapter 2.

Several studies have evaluated the use of the Penman equation for predicting wetland ET (Allen et al., 1992; Faulkner and Lambert, 1991; Koch and Rawlik, 1993; Lafleur, 1990). In most of these studies a crop coefficient (the ratio of wetland ET/reference ET) was developed. Again crop coefficients vary widely and are dependent on site specifics, such as the size of the wetland stand. It has been found that narrow bands of wetland vegetation along streams and lakes or in isolated stands have higher ET rates than wetland vegetation in expansive monotypic stands. This is due to the wind effect ("clothes line effect") which carries away moist air from the plants and increases ET. This wind effect causes the advective losses (the horizontal flux of sensible heat) of energy to become very great and increases the ET (Anderson and Idso, 1987). Anderson and Idso (1987) also showed that as the surface area or canopy increases to a particular size the ET rate decreases. They speculate that the larger surface areas decrease the atmospheric turbulence and decrease advective losses. One must be careful that crop coefficients only be used with the equation they were developed for and with similar site conditions. ET data based on the Penman approach is available in most regions from NOAA climate centers or the U.S. Bureau of Reclamation's AGRIMET Stations. The reported ET from these sites is for a reference crop, usually alfalfa and must be modified by a crop coefficient. The crop coefficient is based on the growing season and maximum ET rates. Table 3.1 is a list of

Table 3.1. Crop coefficients for various hydrophytes and evapotranspiration methods.

Plant/Wetland Type	Location	Crop Coefficient (k_c)	ET Method	Notes	Source
Typha spp.	Utah U.S.A.	1.6	Penman-Monteith	for isolated stands	Allen et al. 1992
Scirpus spp.	Utah U.S.A.	1.8	Penman-Monteith	for isolated stands	Allen et al. 1992
Freshwater Marsh	Florida U.S.A.	1.00 (0.61-2.5)*	Thornwaite	monthly values reported	Dolan et al. 1984
Freshwater Marsh	Florida U.S.A.	0.79 (0.51-1.06)*	Linacre	monthly values reported	Dolan et al. 1984
Freshwater Marsh	Florida U.S.A.	0.67 (0.34-1.16)*	Pan Evaporation	monthly values reported	Dolan et al. 1984
Dambo Marsh	Africa	0.5	Penman FAO-24	grazed grasses & sedges	Faulkner and Lambert, 1991
Sedges (*Carex* spp.)	Ontario, Canada	0.9	Penman Open Water	subartic coastal wetlands	Lafleur, 1990
Taro Field	Florida U.S.A.	0.74-0.95	Pan Evaporation		Shih and Synder, 1984

* values in parentheses indicate range of monthly coefficients

reported crop coefficients for wetlands, the site specifics, and the equation they were used with. Daily ET, crop coefficient, and growing season must be input into this model.

3.5.5 Precipitation

Precipitation, which includes rain, snowfall, and hail, can be a very significant part of a wetland water budget. The amount of precipitation that occurs in a given area is a function of various climatological factors, such as elevation, distance from a large water body, and local topography. Precipitation also follows distinct seasonal patterns in most locations.

The NOAA have weather stations located throughout the United States that publish monthly summary reports. These reports are a convenient method for determining precipitation in a given area. Care must be taken that the NOAA weather stations are not to far from the actual wetland location because some rain events can be extremely localized. Probabilistic models exist for predicting future rainfall events but these are complex and not very accurate. Use of historical data will suffice for most wetland applications.
Daily precipitation is a required input into this model.

3.6 Results and Discussion

A conceptual diagram of the "generic" simulation model was developed from the governing equations discussed above. Figure 3.3 shows the model developed with STELLA™. The model consists of three state variables, and twelve flow pathways. The three state variables are: water stored in the catchment, snow in the catchment, and water in the wetland. Pathways include: runoff from precipitation, direct rainfall, runoff from snowmelt, wastewater loading, streamflow to and from the wetland, infiltration, and evapotranspiration. This "generic" model was modified to fit the specific study site at Oregon State University.

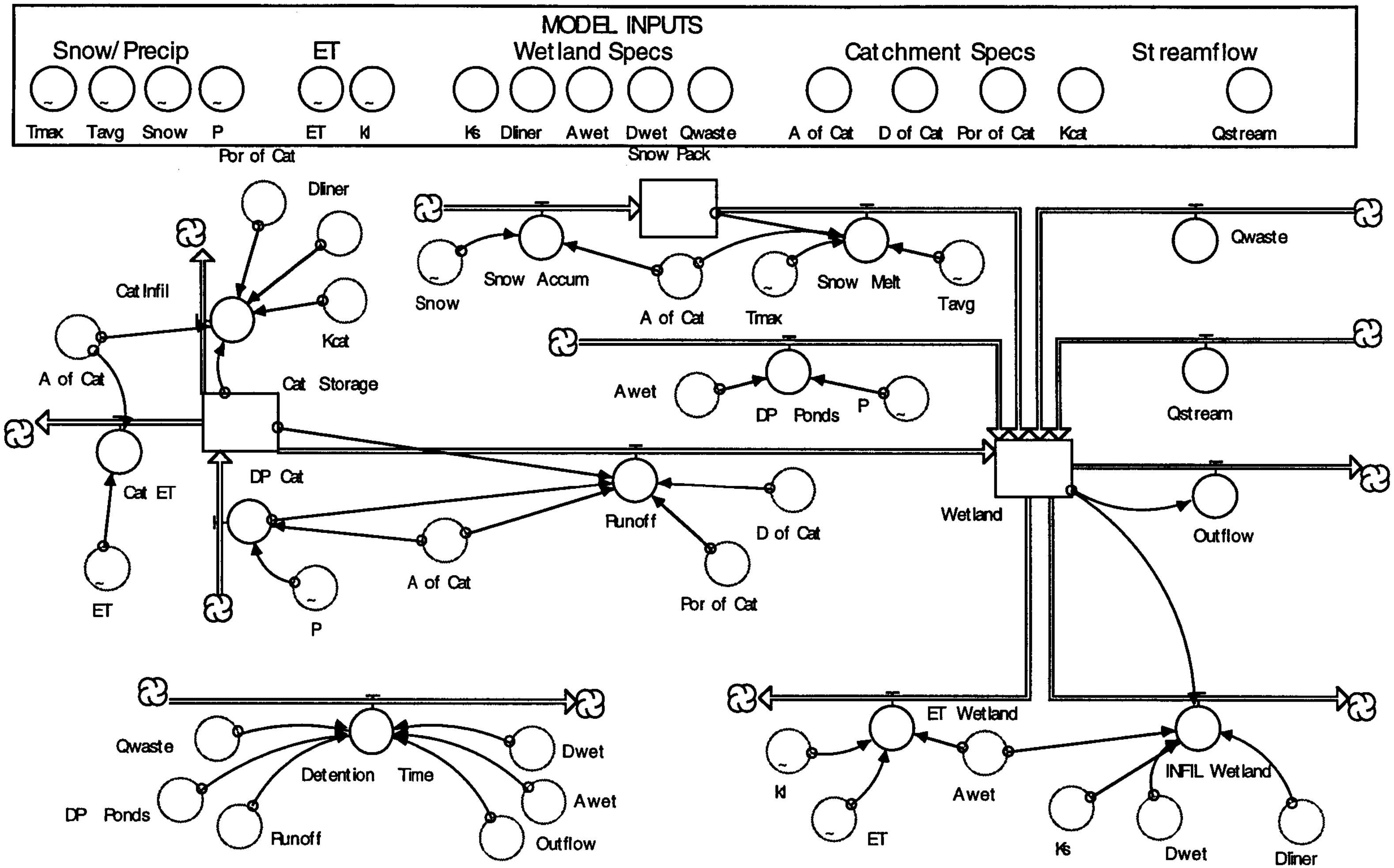

Figure 3.3. Wetland hydrology model developed in STELLA™. Boxes are state variables, circles are functions and parameters, clouds indicate forcing functions and sinks, black lines are flows of energy or material, and gray lines are flows of information. Required inputs to the model are found in the box at the top of the diagram.

The study site is a "closed" system and all outlet water is held and recycled from a storage pond (Fig. 3.1). This required the addition of three state variables: the storage pond (pond 10), the water storage in the berms surrounding pond 10, and the mixing tank where the recycled water and wastewater are mixed. Water is also pumped out of the storage pond during the winter to prevent overflow and water is added in the summer to prevent the storage pond from drying out. Snowfall rarely occurs so the snow pack state variable was removed. There is no streamflow into the wetland cells so this was also removed from the model for this application. The modified STELLA™ model is shown in Figure 3.4. Tables 3.2 and 3.3 summarize the state variables, coefficients, and equations used in the model.

3.6.1 Model Calibration

The model was calibrated against the water depth that was continuously monitored in the storage pond during 1996. This provided an excellent check of the overall water budget. Initial calibrations indicated that the model was under predicting water loss immediately after large rainfall events. Examination of the hydrograph revealed that water was lost at a high rate when water depth was above 68 cm. This water loss rate was found to be in direct relationship to the area of newly wetted bank. The loss rate was very high (1 m^3/m^2 of newly wetted bank-day). This is over 100 times the rate expected for compacted clay. Investigation of the pond banks revealed at least one nutria burrow in the bank. It is believed that nutria burrows, which are as large as 30 cm in diameter, provide "pipes" for water to be lost from pond 10 to the outside of the berms. No other explanations could be found for the rapid water losses following large rainfall events. Water loss through the berms when the water depth was above 68 cm was calculated using an empirically fit equation (see Table 3.3, Q_{berm}). Adding this flow pathway to the model resulted in accurate predictions during the peak storm events.

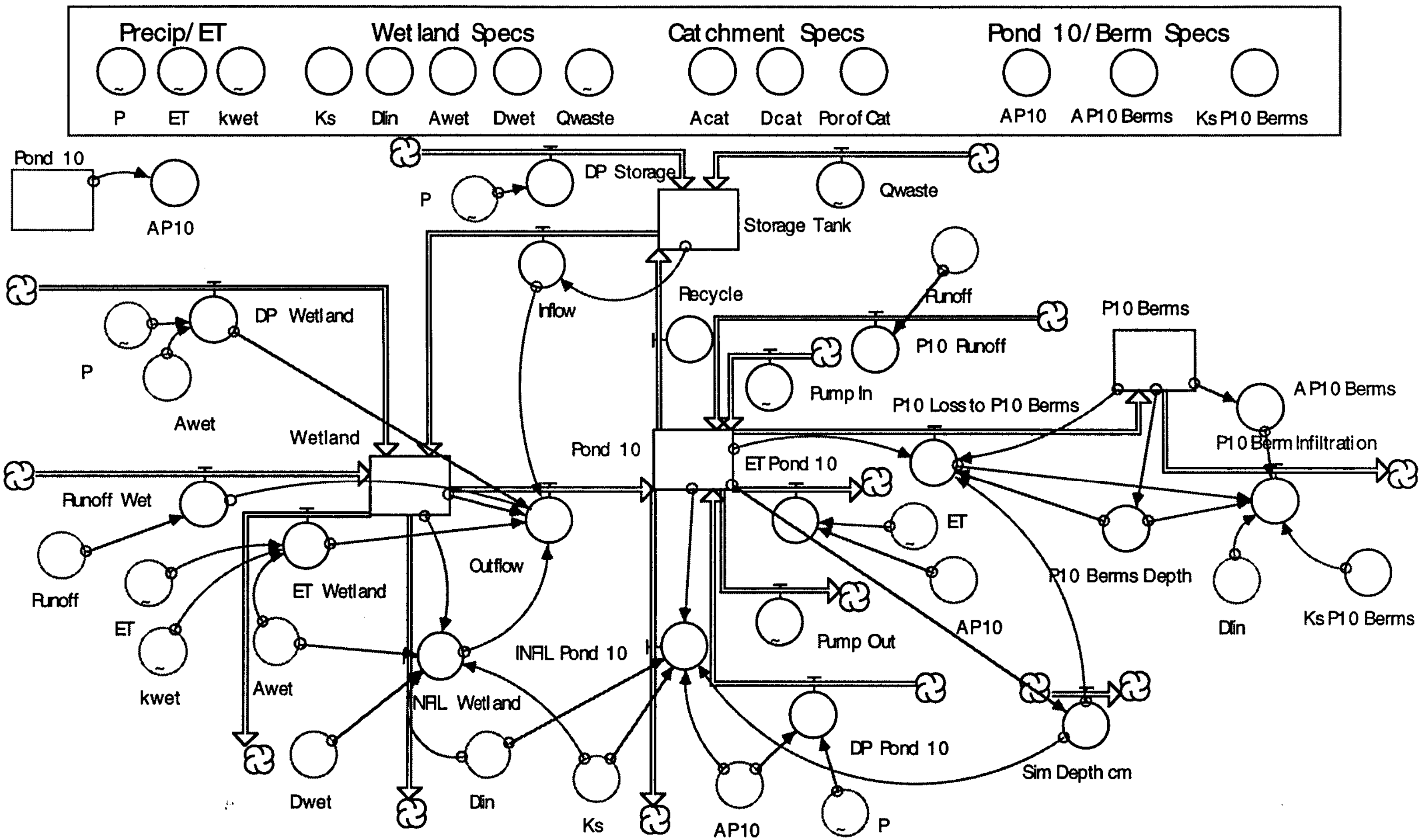

Figure 3.4. Wetland hydrology model for Oregon State University Dairy Wetland Treatment System, Corvallis, OR, developed in STELLA™. Boxes are state variables, circles are functions and parameters, clouds indicate forcing functions and sinks, black lines are flows of energy or material, and gray lines are flows of information. Inputs to the model are found in the box at the top of the diagram.

Table 3.2. Model parameters, definitions, values and sources for the Oregon State University Dairy Wetland Treatment System Hydrology Model.

Symbol	Name	Values/units	Source
State Variables			
V_{wet}	water volume in wetland cells	m^3	field data
V_{10}	volume of pond 10	m^3	field data
V_{berm}	volume of water stored in pond 10 berms	m^3	site data
V_{cat}	water volume stored in wetland catchment	m^3	site data
V_{tank}	volume of water in storage tank	m^3	field data
Forcing Functions			
$P(t)$	precipitation	m/d	AGRIMET data
$ET(t)$	evapotranspiration	m/d	see Chapter 2
$Q_{rc}(t)$	water recycled from 10 to storage tank	m^3/d	field data
$Q_{pin}(t)$	volume of water pumped into pond 10	m^3/d	field data
$Q_{pout}(t)$	volume of water pumped out of pond 10	m^3/d	field data
$Q_{waste}(t)$	volume of wastewater added from dairy	m^3/d	field data
Parameters and Coefficients			
A_{10}	area of pond 10 = $f(V_{10})$	variable (m^2)	field data
A_{cat}	area of catchment	655.2 m^2	field data
A_{tank}	area of tank	14.25 m^2	field data
A_{wet}	area of wetland	955 m^2	field data
D_{10}	depth of pond 10 = $f(V_{10})$	variable (m)	field data
D_{berm}	depth of water in berm soils = $f(V_{berm})$	variable (m)	field data
D_{cat}	depth of water in catchment = $f(V_{cat})$	variable (m)	field data
D_{lin}	thickness of soil liner	38.1 cm	field data
D_{wet}	average wetland water depth	30.5 cm	field data
k_{wet}	crop coefficient for wetland plants	variable	see Chapter 2
k_{cat}	crop coefficient for catchment vegetation	0.8	ASCE, 1990
$k_{cat\%}$	% of catchment that drains into wetland	0.84	field data
k_{10}	crop coefficient for pond 10	0.6	ASCE, 1990
$k_{10cat\%}$	% of catchment that drains into pond 10	0.84	field data
k_s	saturated hydraulic conductivity of soil liner	0.532 mm/d	field data
k_{scat}	saturated hydraulic conductivity of catchment	4.3 mm/d	calibration
k_{sberm}	saturated hydraulic conductivity of berm	0.532 mm/d	field data
k_r	runoff coefficient for rational method	0.9	calibration
k_{nut}	water loss nutria coefficient	0.97	calibration
k_{10berm}	loss rate to berms	1 m^3/m^2-d	calibration
SA_{berm}	area of newly wetted berm = $f(V_{10}, V_{berm})$	m^2	field data
V_{catmax}	max. water volume stored in catchment	m^3	site data
V_{wetmax}	max. water volume stored before outflow	m^3	field data

Table 3.3. State variables and differential equations for the Oregon State University Dairy Wetland Treatment System Hydrology Model.

Water in Wetland Cells, V_{wet}

$dV_{wet}/dt = Q_{in}(t) + P(t)\cdot A_{wet} + Q_r(t)\cdot k_{cat\%} - Q_g(t) - Q_{out}(t) - ET(t)\cdot A_{wet}\cdot k_{wet}$

where,	V_{wet}	water volume in wetland (m^3)
	$Q_{in}(t)$	mixed wastewater inflow rate = V_{tank} (m^3/d)
	$P(t)$	precipitation (m/day)
	A_{wet}	area of wetland (m^2)
	$Q_r(t)$	runoff (m^3/d) = $f(V_{cat})$
	$k_{cat\%}$	percentage of catchment that drains into wetland cells (%)
	$Q_g(t)$	wetland cells infiltration (m^3/d) = $k_s\cdot A_{wet}\cdot(D_{wet} + D_{lin})/(D_{lin})$
	k_s	saturated hydraulic conductivity of soil liner (m/d)
	D_{wet}	average wetland water depth (m)
	D_{lin}	thickness of soil liner (m)
	$Q_{out}(t)$	outflow (m^3/d)= $f(V_{wet})$ = If($V_{wet} > V_{wetmax}$) Then ($V_{wet} - V_{wetmax}$) Else (0)
	V_{wetmax}	water volume that can be stored in the wetland before outflow (m^3)
	$ET(t)$	evapotranspiration (m/d)
	k_{wet}	crop coefficient for wetland plants

Water in Catchment Soil, V_{cat}

$dV_{cat}/dt = P(t)\cdot A_{cat} - Q_{catg}\cdot(t) - ET(t)\cdot A_{cat}\cdot k_{cat} - Q_r(t)$

where,	V_{cat}	water volume stored in wetland catchment (berms) (m^3)
	A_{cat}	area of catchment (m^2)
	$Q_{catg}(t)$	catchment infiltration (m^3/d) = $k_{cat}\cdot A_{wet}\cdot(D_{cat} + D_{lin})/(D_{lin})$
	k_{scat}	saturated hydraulic conductivity of soil in catchment (m/d)
	D_{cat}	depth of water in catchment (m) = $f(V_{cat})$
	$ET(t)$	evapotranspiration (m/d)
	k_{cat}	crop coefficient for catchment vegetation
	$Q_r(t)$	runoff (m3/d) = IF[($V_{cat} > V_{catmax}$) AND (90 > Day > 270)] THEN ($V_{cat} - V_{catmax}$) ELSE ($k_r\cdot P(t)\cdot A_{cat}$)
	V_{catmax}	maximum water volume that can be stored in catchment (m^3)
	k_r	runoff coefficent for rational method

Table 3.3. (continued)

Water in Storage Pond (Pond 10), V_{10}

$$dV_{10}/dt = Q_{out}(t) + P(t)\cdot A_{10} + Q_{pin} + Q_r(t)\cdot k_{10cat\%} - Q_{pout} - Q_{10g} - Q_{rc} - ET(t)\cdot A_{10}\cdot k_{10} - Q_{berm}$$

where,	V_{10}	volume of pond 10 (m^3)
	A_{10}	area of pond 10 (m^3) = $f(V_{10})$
	Q_{pin}	volume of water pumped into pond 10 (m^3/d)
	$Q_r(t)$	runoff (m^3/d) = $f(V_{cat})$
	$k_{10cat\%}$	percentage of catchment that drains into pond 10 (%)
	Q_{pout}	volume of water pumped out of pond 10 (m^3/d)
	Q_{10g}	pond 10 infiltration (m^3/d) = $k_{wet}\cdot A_{10}\cdot(D_{10} + D_{lin})/(D_{lin})$
	D_{10}	depth of pond 10 = $f(V_{10})$
	Q_{rc}	volume of water recycled to storage tank (m^3/d)
	k_{10}	crop coefficient for pond 10
	Q_{berm}	water loss to berms (m^3/d) = IF($D_{10} > D_{berm}$) AND ($D_{10} > 0.68$) THEN ($k_{10berm}\cdot SA_{berm}$) ELSE (0)
	k_{10berm}	loss rate to berms (m^3/m^2-d)
	SA_{berm}	surface area of newly wetted berm (m^2)

Water in Pond 10 Berms, V_{berm}

$$dV_{berm}/dt = Q_{berm} - Q_{bermg} - Q_{nutria}$$

where,	V_{berm}	volume of water stored in pond 10 berms (m^3)
	Q_{bermg}	pond 10 berm infiltration (m^3/d) = $k_{sberm}\cdot A_{berm}\cdot(D_{berm} + D_{lin})/(D_{lin})$)
	k_{sberm}	saturated hydraulic conductivity of berm (m/d)
	D_{berm}	depth of water in berm soils (m) = $f(V_{berm})$
	Q_{nutria}	volume of water loss through nutria burrow (m^3/d) = IF ($D_{berm} >$ 0.68) THEN ($k_{nut}\cdot Q_{berm}$) ELSE (0)
	k_{nut}	water loss nutria coefficent (unitless)

Water in Storage Tank, V_{tank}

$$dV_{tank}/dt = Q_{rc} + P(t)A_{tank} + Q_{waste}(t) - Q_{in}(t)$$

where,	V_{tank}	volume of water in storage tank (m^3)
	A_{tank}	area of tank (m^2)
	Q_{waste}	volume of concentrated wastewater added from dairy (m^3/d)

Figure 3.5 shows the simulated versus actual water depth in pond 10 for 1996. The model does a very good job of predicting water levels throughout the year (R^2 = 0.95). A scatterplot of actual versus simulated depth shows that no outliers exist and the model is accurate at all depths (Fig. 3.6). Table 3.4 shows the calibrated monthly water budget for the entire system. It is apparent from Figure 3.5 that high water levels occur throughout the winter months and low water levels during the summer. This is a result of the climatic conditions of Corvallis, OR, which receives an average of 80% of its annual 108 cm of rain between October and March (Taylor and Bartlett, 1993). This corresponds to the period of lowest evapotranspiration. As a result of the high rainfall and low evapotranspiration during this period, it is often necessary to pump water out of pond 10 to prevent overflowing. The opposite conditions exist during the summer: low rainfall and high evapotranspiration. This causes extremely low water levels in pond 10 and water must be added throughout the summer months to maintain enough water for recycling.

Table 3.5 is the 1996 water budget of just the wetland treatment cells. Recycled water loading dominates the overall water budget. However, the seasonal rainfall and evapotranspiration patterns cause shorter detention times in the winter and longer in the summer (Table 3.5). During the winter, the detention time based on the monthly average is up to 19% shorter than the theoretical detention time and up to 9% longer during summer. For a shorter time interval, the detention time can deviate by even a greater amount. The rainfall also causes a dilution effect in the winter and the high evapotranspiration concentrates the wastewater in the summer. Both the change in detention times and the dilution or concentration effects can have significant effects on the wastewater treatment. When evaluating the treatment performance of a constructed wetland, it is critical that the water budget be taken into account.

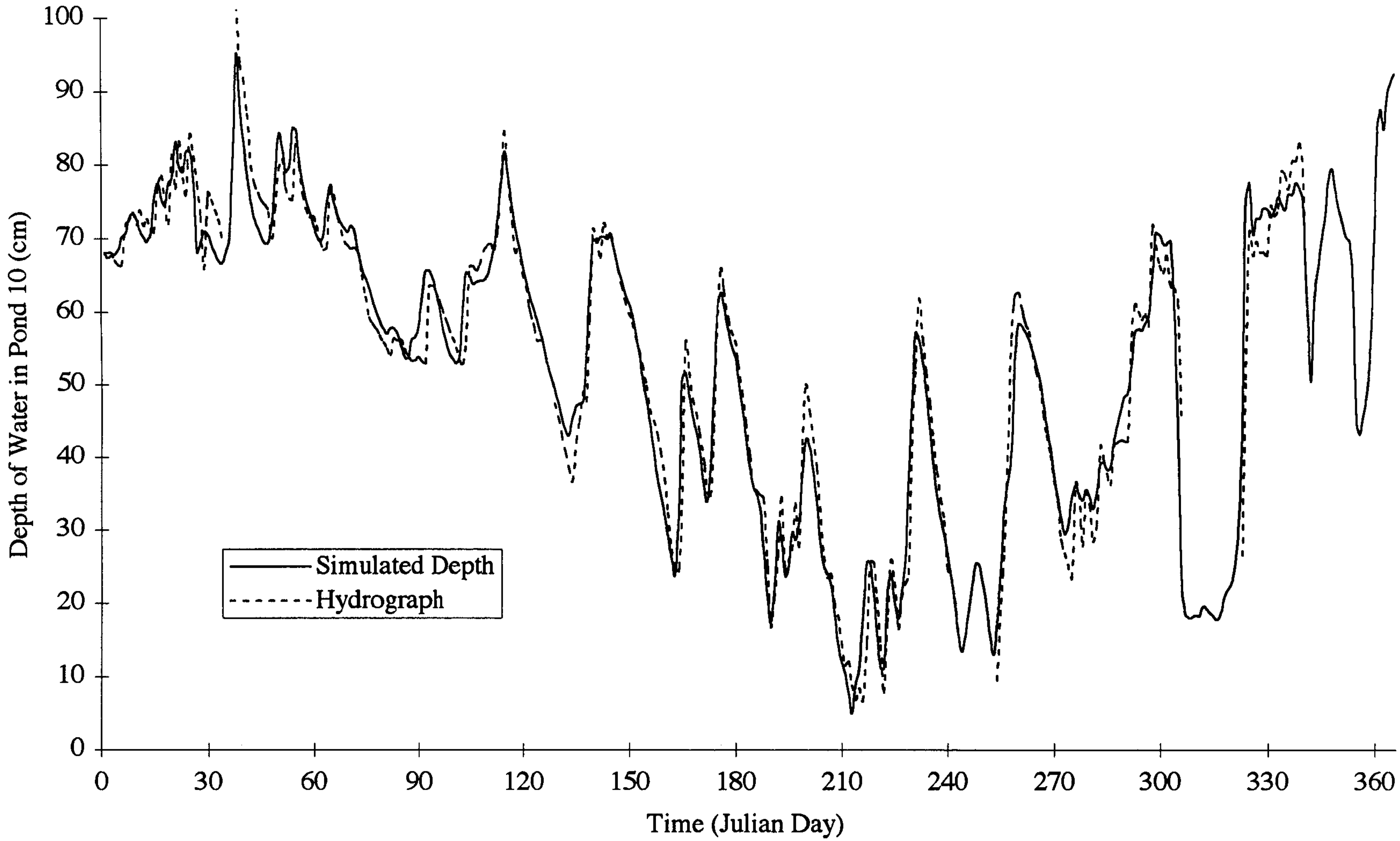

Figure 3.5. 1996 simulated versus actual water depth in Pond 10.

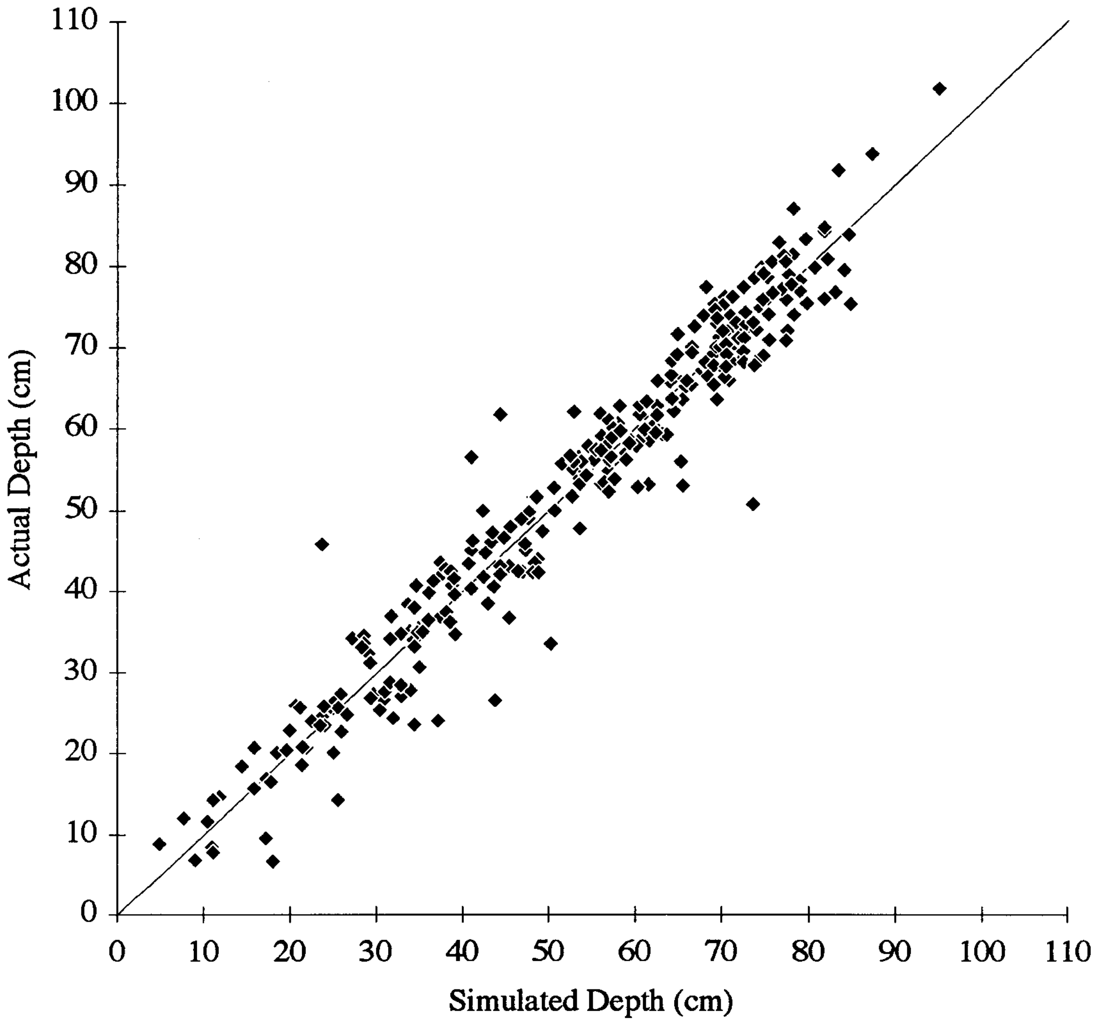

Figure 3.6. Scatterplot of daily actual depth versus simulated depth. This indicates calibration for the model ($R^2 = 0.95$).

Table 3.4. Calibrated 1996 monthly water budget for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precipitation	Runoff	Wastewater Loading	Water Pumped Into Pond 10	Water Pumped Out of Pond 10	Evapo-transpiration	Infiltration	Berm Loss (Nutria Burrows)
Jan	360	62	12	0	-179	-23	-47	-183
Feb	475	130	11	0	-212	-61	-44	-291
Mar	130	10	12	0	-9	-81	-45	-40
Apr	163	37	11	20	0	-123	-43	-68
May	136	20	12	40	0	-172	-43	-17
Jun	29	1	11	236	0	-274	-40	0
Jul	31	5	4	269	0	-384	-38	0
Aug	5	0	12	384	0	-343	-38	0
Sep	75	19	11	193	0	-206	-38	0
Oct	182	0	10	63	-117	-83	-42	-13
Nov	360	0	4	0	-75	-20	-40	-88
Dec	580	180	12	0	-368	-15	-46	-285
Annual	2,525	463	122	1,205	-960	-1,785	-504	-984
% of Total Inflow	58.5%	10.7%	2.8%	27.9%	-22.3%	-41.4%	-11.7%	-22.8%

Table 3.5. Calibrated 1996 monthly water budget and retention times for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precip.	Runoff	Recycled Water Loading	Wastewater Loading	Mixed Wastewater Loading[1]	Outflow	Evapo-transpiration	Infiltration	Average Retention Time (RT) (days)[2]	Deviation from Theoretical R.T.[3]
Jan	266	52	2145	12	2157	-2426	-19	-29	3.91	-10%
Feb	349	109	2007	11	2018	-2398	-50	-28	3.85	-12%
Mar	97	8	2145	12	2157	-2165	-68	-29	4.27	-2%
Apr	121	31	2076	11	2087	-2105	-106	-28	4.24	-3%
May	102	17	2145	12	2157	-2095	-151	-29	4.33	-1%
Jun	22	1	2076	11	2087	-1837	-245	-28	4.61	6%
Jul	24	4	2145	4	2149	-1801	-346	-29	4.74	9%
Aug	4	0	2145	12	2157	-1823	-308	-29	4.72	8%
Sep	57	16	2076	11	2087	-1947	-185	-28	4.47	3%
Oct	137	0	2145	10	2156	-2190	-73	-29	4.22	-3%
Nov	270	0	2076	4	2080	-2304	-17	-28	4.02	-8%
Dec	430	151	2145	12	2157	-2695	-13	-29	3.58	-18%
Annual	1905	389	25327	122	25449	-25787	-1580	-348	4.25	-2.6%
% of Total Inflow	6.9%	1.4%	91.3%	0.4%	91.7%	-93.0%	-5.7%	-1.3%		

[1] Mixed wastewater loading = recycled water loading (Q_{rc}) + concentrated wastewater loading (Q_{waste}).

[2] Average of daily calculated retention times.

[3] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

3.6.2 Simulations

The calibrated model was slightly modified so it could be used to simulate the water budget for any year. This modification was to change the pumping of freshwater to and from pond 10 from a required input into a logical statement:

If (Pond 10 Depth (D_{10}) > 80 cm) Then (Pump Out (Q_{pout}) = 104 m^3/d)) Else (0),

If (Pond 10 Depth (D_{10}) < 15 cm) Then (Pump In (Q_{pin}) = 104 m^3/d)) Else (0).

This modified model was used to simulate 1996 conditions and resulted in an overall annual water budget similar to the budget predicted by the calibrated model (Table 3.6). In the revised model, less water was pumped into out of pond 10 and more water was lost through the berms, 101, 125, and 64 m^3, respectively. In the winter, the water was deeper in pond 10, therefore, the surface area was larger and direct precipitation into pond 10 was slightly greater. In addition, greater infiltration occurred in pond 10 as a result of the increased depth. Lower depths in the summer resulted in decreased infiltration and a decreased surface area, which lowered the evapotranspiration loss. The revised model and calibrated 1996 models predicted identical water budgets for the wetland cells.

3.6.2.1 Effect of Annual Rainfall and Evapotranspiration

The revised model was used to simulate the water budget for 1994, 1995, and 1996. The annual rainfall for 1994, 1995, and 1996 were 98 cm, 138, and 186 cm. The average annual rainfall for Corvallis, OR is 108 cm (Taylor and Bartlett, 1993). It is interesting to note that 1996 was the wettest calendar year in recorded history and surpassed the previous record by over 36 cm. Table 3.7 is a summary of the annual water budgets for all three simulations. Tables 3.8, 3.9, and 3.10 are the monthly water budgets for just the wetland cells. For the calendar years of 1994 (an "average" rainfall year), 1995 (a wet year), and 1996 (the wettest year in recorded history) direct precipitation plus runoff

Table 3.6. 1996 monthly water budget for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR, with automatic pumping of water into and out of pond 10. Calibrated 1996 annual water budget is shown at bottom. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precipitation	Runoff	Wastewater Loading	Water Pumped Into Pond 10	Water Pumped Out of Pond 10	Evapo-transpiration	Infiltration	Berm Loss (Nutria Burrows)
Jan	361	62	12	0	-61	-23	-47	-275
Feb	477	130	11	0	-307	-61	-45	-214
Mar	131	10	12	0	0	-82	-46	-41
Apr	163	37	11	0	0	-124	-44	-66
May	136	20	12	0	0	-171	-43	0
Jun	28	1	11	182	0	-272	-36	0
Jul	31	5	4	380	0	-383	-37	0
Aug	5	0	12	361	0	-342	-37	0
Sep	74	19	11	139	0	-205	-37	0
Oct	179	0	10	43	0	-82	-39	0
Nov	366	0	4	0	-71	-20	-44	-161
Dec	586	180	12	0	-397	-15	-48	-291
Auto-Pump Annual	2,536	463	122	1,104	-835	-1,780	-503	-1,048
% of Total Inflow	60.0%	11.0%	2.9%	26.1%	-19.8%	-42.1%	-11.9%	-24.8%
Calibrated Annual	2,525	463	122	1,205	-960	-1,785	-504	-984
% of Total Inflow	58.5%	10.7%	2.8%	27.9%	-22.2%	-41.4%	-11.7%	-22.8%

Table 3.7. 1994, 1995, and 1996 annual water budgets for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Depth = 30.48 cm. Positive values indicate water additions and negative values are water losses.

Year		Direct Precipitation	Runoff	Wastewater Loading	Water Pumped Into Pond 10	Water Pumped Out of Pond 10	Evapo-transpiration	Infiltration	Berm Loss (Nutria Burrows)
1994	m^3	1,320	24	139	1,605	-10	-2,043	-486	-514
	%	42.7%	0.8%	4.5%	52.0%	-0.3%	-66.2%	-15.7%	-16.6%
1995	m^3	1,875	146	139	1,225	-161	-1,875	-498	-812
	%	55.4%	4.3%	4.1%	36.2%	-4.8%	-55.4%	-14.7%	-24.0%
1996	m^3	2,536	463	139	1,097	-839	-1,780	-503	-1,054
	%	59.9%	10.9%	3.3%	25.9%	-19.8%	-42.0%	-11.9%	-24.9%

Table 3.8. 1994 monthly water budget and retention times for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Depth = 30.48 cm. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precip.	Runoff	Recycled Water Loading	Wastewater Loading	Mixed Wastewater Loading[1]	Outflow	Evapo-transpiration	Infiltration	Average Retention Time (RT) (days)[2]	Deviation from Theoretical R.T.[3]
Jan	95	0	2145	12	2157	-2185	-37	-29	4.26	-2%
Feb	144	0	1938	11	1948	-2024	-41	-27	4.16	-5%
Mar	89	5	2145	12	2157	-2139	-83	-29	4.30	-1%
Apr	50	2	2076	11	2087	-1983	-128	-28	4.42	1%
May	31	1	2145	12	2157	-1936	-223	-29	4.56	5%
Jun	46	3	2076	11	2087	-1852	-256	-28	4.57	5%
Jul	0	0	2145	12	2157	-1742	-386	-29	4.82	11%
Aug	0	0	2145	12	2157	-1825	-302	-29	4.72	8%
Sep	23	0	2076	11	2087	-1848	-234	-28	4.60	6%
Oct	130	0	2145	12	2157	-2160	-97	-29	4.29	-1%
Nov	217	0	2076	11	2087	-2259	-17	-28	4.03	-8%
Dec	162	10	2145	12	2157	-2285	-15	-29	4.11	-6%
Annual	1000	20	25258	139	25397	-24237	-1818	-347	4.40	1.0%
% of Total Inflow	3.8%	0.1%	95.6%	0.5%	96.1%	-91.7%	-6.9%	-1.3%		

[1] Mixed wastewater loading = recycled water loading (Q_{rc}) + concentrated wastewater loading (Q_{waste}).

[2] Average of daily calculated retention times.

[3] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

Table 3.9. 1995 monthly water budget and retention times for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Depth = 30.48 cm. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precip.	Runoff	Recycled Water Loading	Wastewater Loading	Mixed Wastewater Loading[1]	Outflow	Evapo-transpiration	Infiltration	Average Retention Time (RT) (days)[2]	Deviation from Theoretical R.T.[3]
Jan	258	39	2145	12	2157	-2405	-19	-29	3.96	-9%
Feb	106	7	1938	11	1948	-2002	-32	-27	4.23	-3%
Mar	126	1	2145	12	2157	-2185	-69	-29	4.23	-3%
Apr	137	39	2076	11	2087	-2126	-109	-28	4.18	-4%
May	34	2	2145	12	2157	-1928	-235	-29	4.57	5%
Jun	61	9	2076	11	2087	-1860	-268	-28	4.56	5%
Jul	13	1	2145	12	2157	-1819	-323	-29	4.72	8%
Aug	21	0	2145	12	2157	-1850	-299	-29	4.67	7%
Sep	80	6	2076	11	2087	-1951	-194	-28	4.46	2%
Oct	102	0	2145	12	2157	-2142	-87	-29	4.31	-1%
Nov	210	0	2076	11	2087	-2249	-19	-28	4.04	-7%
Dec	248	20	2145	12	2157	-2383	-13	-29	3.97	-9%
Annual	1415	123	25258	139	25397	-24901	-1667	-347	4.32	-0.8%
% of Total Inflow	5.3%	0.5%	93.8%	0.5%	94.3%	-92.4%	-6.2%	-1.3%		

[1] Mixed wastewater loading = recycled water loading (Q_{rc}) + concentrated wastewater loading (Q_{waste}).

[2] Average of daily calculated retention times.

[3] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

Table 3.10. 1996 monthly water budget and retention times for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Depth = 30.48 cm. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precip.	Runoff	Recycled Water Loading	Wastewater Loading	Mixed Wastewater Loading[1]	Outflow	Evapo-transpiration	Infiltration	Average Retention Time (RT) (days)[2]	Deviation from Theoretical R.T.[3]
Jan	266	52	2145	12	2157	-2426	-19	-29	3.91	-10%
Feb	349	109	2007	11	2018	-2398	-50	-28	3.85	-12%
Mar	97	8	2145	12	2157	-2165	-68	-29	4.27	-2%
Apr	121	31	2076	11	2087	-2105	-106	-28	4.24	-3%
May	102	17	2145	12	2157	-2095	-151	-29	4.33	-1%
Jun	22	1	2076	11	2087	-1837	-245	-28	4.61	6%
Jul	24	4	2145	12	2157	-1809	-346	-29	4.72	8%
Aug	4	0	2145	12	2157	-1823	-308	-29	4.72	8%
Sep	57	16	2076	11	2087	-1947	-185	-28	4.47	3%
Oct	137	0	2145	12	2157	-2191	-73	-29	4.22	-3%
Nov	270	0	2076	11	2087	-2312	-17	-28	4.01	-8%
Dec	430	151	2145	12	2157	-2695	-13	-29	3.58	-18%
Annual	1905	389	25327	139	25466	-25804	-1580	-348	4.24	-2.7%
% of Total Inflow	6.9%	1.4%	91.2%	0.5%	91.7%	-93.0%	-5.7%	-1.3%		

[1] Mixed wastewater loading = recycled water loading (Q_{rc}) + concentrated wastewater loading (Q_{waste}).

[2] Average of daily calculated retention times.

[3] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

accounted for 3.9%, 5.8%, and 8.3% of the total inflow, respectively. Evapotranspiration accounted for 6.9%, 6.2%, and 5.7% of the total water loss in 1994, 1995, and 1996. This is a fairly small proportion of the overall water budget and empirical estimates of ET would probably be accurate enough. However, if the hydraulic loading rates were lower, the importance of ET in the overall water budget would be much greater and accurate calculations of ET would be required. The effect of the water additions in the winter was to decrease the actual detention time by an average of 3.8%, 5.3%, and 8.8% for 1994, 1995, and 1996. Evapotranspiration and infiltration losses in the summer resulted in longer detention times than the theoretical detention time. For 1994, 1995, and 1996 detention times were and average of 6.0%, 3.8%, and 3.5% longer during the summer months. The change in detention times during the summer and winter seasons coupled with the dilution and concentration could have a dramatic effect on treatment performance.

3.6.2.2 Effect of Wastewater Loading Rate

If the wastewater hydraulic loading rate decreases the effect of evapotranspiration and precipitation on the detention time is even greater. Table 3.11 is the monthly water budget for 1996 when the wastewater flow rate is adjusted to achieve a theoretical detention time of 8 days. Comparing Tables 3.10 and 3.11, one can see that if the mixed wastewater loading rate is decreased than the importance of the other flow paths becomes more important. The deviation of the detention time from theoretical also increases dramatically when the mixed wastewater loading rate is decreased. In general, the importance of the other hydrologic functions decreases as the detention time decreases (i.e. as the wastewater hydraulic loading rate increases.

Table 3.11. 1996 monthly water budget and retention times for wetland cells at Oregon State University Dairy Wetland Treatment System, Corvallis, OR, with an 8 day retention time. Depth = 30.48 cm. All values are in m^3. Positive values indicate water additions and negative values are water losses.

Month	Direct Precip.	Runoff	Recycled Water Loading	Wastewater Loading	Mixed Wastewater Loading[1]	Outflow	Evapo-transpiration	Infiltration	Average Retention Time (RT) (days)[2]	Deviation from Theoretical R.T.[3]
Jan	266	52	1163	12	1175	-1476	-19	-29	6.57	-18%
Feb	349	109	1088	11	1099	-1480	-50	-28	6.70	-16%
Mar	97	8	1163	12	1175	-1183	-68	-29	7.75	-3%
Apr	121	31	1126	11	1137	-1155	-106	-28	7.69	-4%
May	102	17	1163	12	1175	-1113	-151	-29	7.97	0%
Jun	22	1	1126	11	1137	-886	-245	-28	8.93	12%
Jul	24	4	1163	12	1175	-827	-346	-29	9.38	17%
Aug	4	0	1163	12	1175	-841	-308	-29	9.33	17%
Sep	57	16	1126	11	1137	-997	-185	-28	8.48	6%
Oct	137	0	1163	12	1175	-1209	-73	-29	7.60	-5%
Nov	270	0	1126	11	1137	-1361	-17	-28	7.01	-12%
Dec	430	151	1163	12	1175	-1713	-13	-29	5.88	-27%
Annual	1905	389	13732	139	13871	-14241	-1580	-348	7.77	-2.8%
% of Total Inflow	11.8%	2.4%	85.0%	0.9%	85.8%	-88.1%	-9.8%	-2.2%		

[1] Mixed wastewater loading = recycled water loading (Q_{rc}) + concentrated wastewater loading (Q_{waste}).

[2] Average of daily calculated retention times.

[3] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

3.6.2.3 Effect of Depth and Surface Area

A series of simulations was carried out to investigate the effect of wetland surface area and depth on the overall water budget. In all of these simulations, the size of pond 10, the size of the catchment, and the volume of the wetland cells were held constant. Because the wetland volume was held constant, any increase in wetland depth resulted in a proportional decrease in wetland surface area and visa versa. Five simulations were carried out with wetland depths at 7.62, 15.24, 30.48, 45.72, and 60.96 cm. Table 3.12 is the annual water budgets for the wetland cells for each simulation. Holding the wetland volume constant, the surface area decreases as depth increases, therefore, both direct precipitation and evapotranspiration decrease. During the winter months, runoff is based on the storage capacity of the berms (Table 3.3). As wetland depth increases, the storage capacity of the berm decreases because it is assumed the berms remain saturated to the depth of the wetland. Thus, as wetland depth increases, the runoff from the catchment increases. The volume of water lost by infiltration decreases as wetland depth increases because the surface area of the water soil interface decreases. However, the rate of infiltration increases slightly due to the increased head. Detention time is dramatically effected by changes in depth and surface area (Table 3.13). While the range of the average annual detention time deviation is only -2.7% to 2.2%, the monthly average deviation is large. In December, the deviation is -36% for 7.62 cm, and -13% for 60.96 cm. In July, the deviation is 50% and 4% for 7.62 cm and 60.96 cm, respectively. Increasing the water depth and decreasing the surface area is one way to minimize the water losses and gains, which minimizes the deviation from the theoretical detention time.

3.7 Conclusions

As shown with the simulations OSUDWTS, seasonal patterns of rainfall and evapotranspiration can result in large fluctuations in the significance of the hydrologic

Table 3.12. 1996 annual water budgets for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR, with various depths. Positive values indicate water additions and negative values are water losses.

Depth (cm)		Direct Precipitation	Runoff	Wastewater Loading	Water Pumped Into Pond 10	Water Pumped Out of Pond 10	Evapo-transpiration	Infiltration	Berm Loss (Nutria Burrows)
7.62	m^3	8,116	414	139	4,718	-3,822	-6,518	-1,084	-1,907
	%	60.6%	3.1%	1.0%	35.2%	-28.6%	-48.7%	-8.1%	-14.2%
15.24	m^3	4,394	424	139	2,269	-1,763	-3,358	-695	-1,353
	%	60.8%	5.9%	1.9%	31.4%	-24.4%	-46.5%	-9.6%	-18.7%
30.48	m^3	2,536	463	139	1,097	-839	-1,780	-503	-1,054
	%	59.9%	10.9%	3.3%	25.9%	-19.8%	-42.0%	-11.9%	-24.9%
45.72	m^3	1,916	501	139	716	-549	-1,255	-439	-968
	%	58.6%	15.3%	4.3%	21.9%	-16.8%	-38.3%	-13.4%	-29.6%
60.96	m^3	1,606	548	139	527	-423	-992	-407	-938
	%	57.0%	19.4%	4.9%	18.7%	-15.0%	-35.2%	-14.4%	-33.3%

Table 3.13. 1996 average monthly retention times and deviation from theoretical for various depths.

	Average Retention Time[1] - Deviation from Theoretical Retention Time[2]									
	7.62 cm		15.24 cm		30.48 cm		45.72 cm		60.96 cm	
Month	(days)	(%)	(days)	(%)	(days)	(%)	(days)	(%)	(days)	(%)
Jan	3.31	-24%	3.66	-16%	3.91	-10%	4.01	-8%	4.05	-7%
Feb	3.47	-20%	3.66	-16%	3.85	-12%	3.93	-10%	3.99	-9%
Mar	4.16	-5%	4.21	-3%	4.27	-2%	4.30	-1%	4.31	-1%
Apr	4.20	-4%	4.19	-4%	4.24	-3%	4.26	-2%	4.27	-2%
May	4.48	3%	4.36	0%	4.33	-1%	4.33	-1%	4.33	-1%
Jun	5.65	29%	4.90	12%	4.61	6%	4.53	4%	4.49	3%
Jul	6.53	50%	5.18	19%	4.72	8%	4.59	5%	4.53	4%
Aug	6.29	44%	5.14	18%	4.72	8%	4.60	5%	4.54	4%
Sep	5.12	18%	4.65	7%	4.47	3%	4.42	1%	4.40	1%
Oct	4.02	-8%	4.12	-6%	4.22	-3%	4.26	-2%	4.28	-2%
Nov	3.46	-21%	3.78	-13%	4.01	-8%	4.07	-7%	4.10	-6%
Dec	2.78	-36%	3.24	-26%	3.58	-18%	3.72	-15%	3.80	-13%
Annual	4.46	2.2%	4.26	-2.4%	4.24	-2.7%	4.25	-2.5%	4.26	-2.3%

[1] Average of daily calculated retention times.

[2] Theoretical retention time based on mixed wastewater loading rate and volume of wetlands = 4.36 days.

functions. These fluctuations can cause either a concentration or dilution effect and also alter the detention time. High rainfall dilutes the wastewater but also decreases the detention time and may decrease the treatment performance. High evapotranspiration rates cause a concentration effect but also increase the detention time, which should result in increased treatment performance.

A water budget is required for calculating any contaminant mass balance, which is the basis for determining treatment performance. Therefore, it is critical that a water budget be calculated for any wetland treatment system. This model provides a tool for easily calculating a detailed water budget for any constructed wetland. It is easily modified to calculate the water budget that matches the frequency of sampling. It also can be used to evaluate the effect of different wetland configurations and designs on the overall water budget. Finally, the model is the foundation for any additional "treatment" submodels.

3.8 References

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4. Hydraulics of a Constructed Wetland Treating Dairy Wastewater

4.1 Abstract

Wastewater treatment in constructed wetlands is dependent on the kinetics and contacting pattern. The contacting pattern describes the duration of contact between the wastewater constituent and the reaction site. The kinetics is the rate at which a reaction occurs. Most studies of constructed wetlands have focused on measuring the kinetics without regard to hydraulics. The most common method for determining the contact pattern is to calculate the theoretical detention time and assume that plug flow exists. Plug flow rarely occurs in wetlands and has resulted in a wide variety of kinetic coefficients being reported.

Tracer studies can be used to determine the actual detention time and flow characteristics. Tracer studies were conducted at the OSUDWTS using Rhodamine WT. It was found that mean detention time was an average of 43% shorter than the theoretical detention time. The residence time distributions were characterized by early peaks and long trailing curves. This indicates a large amount of dead space and/or sorption and desorption of the tracer. The tank in series and plug flow modified by dispersion models were fit to the data but did not adequately describe the flow conditions due to the large amount of dead space. Inadequate flow distribution at the inlet and collection at the outlet may explain the deviation from ideal flow. Use of the theoretical detention time predicted 15% higher removal than the removal predicted with the mean detention time.

Keywords: plug flow; tank in series; dispersion; residence time distribution

4.2 Introduction

Natural and constructed wetlands have been used for treatment of wastewater for over two decades (Kadlec and Knight, 1996). The number of constructed wetlands has increased dramatically in the past few years. The reported treatment efficiencies for these systems vary widely. Databases for constructed wetlands treating industrial/domestic and livestock wastewater in North America have been developed (NADB, 1993; Knight et al., 1996). These report treatment efficiencies that range from 20 to 90% for specific wastewater constituents. While a portion of this variability is likely caused by climatic, design, vegetation, and loading differences, a large part of this difference is probably caused by the failure to take into account the hydraulics of the wetland system.

An understanding of the hydraulics of the wetland is needed for accurately predicting treatment efficiency. Treatment of pollutants in wetlands is based on both kinetics and contacting pattern. Kinetics describe how fast a reaction or conversion occurs and the contacting pattern describes the occurrence and duration of contact between the pollutant and reaction sites. The kinetics in wetlands have received the most attention, often with a disregard for the contacting pattern (hydraulics). The most common method for describing the contacting pattern is to determine the theoretical residence time (theoretical detention time) of the wetland. The theoretical detention time is found by dividing the volume of the wetland by the inlet flow rate. It is then assumed that all particles of water spend an equal amount of time (the theoretical detention time) in the wetland and move uniformly from the inlet to the outlet. This describes the plug flow case which has been widely presumed to occur in constructed wetlands. Numerous studies have shown this assumption to be incorrect (Eberdorfer, 1993, Fisher, 1990; Kadlec and Knight, 1996; Stairs and Moore, 1993). However, all design manuals to date use this assumption (Kadlec and Knight, 1996).

Most attempts at describing flow patterns in free water constructed wetlands have used models developed for open channel flow. These models generally use mass, energy, and momentum conservation equations with an equation to account for frictional resistance (Kadlec and Knight, 1996). The most common friction equation used to describe the resistance in free water surface wetlands is Manning's equation (Reed et al., 1995). However, Kadlec and Knight (1996) argue Manning's equation and open channel flow models are inadequate and not applicable to constructed wetlands. This argument is based on the fact that Manning's equation is a correlation for turbulent flow and flow in wetlands is usually transitional (Kadlec and Knight, 1996). In addition, frictional resistance in open channels is mainly exerted by the drag exerted by the channel sides and bottom (Kadlec and Knight, 1996). Conversely, wetland frictional resistance is largely due to the drag exerted by macrophyte stems and litter.

A different technique that is gaining more use for describing flow patterns in wetlands is to determine the residence time distribution (RTD). The theory of RTDs has been thoroughly discussed in the works of Levenspiel (1972, 1993) and Fogler (1992). The RTD is the distribution of times that various fractions of fluid remain in the reactor, which in this case is the wetland. The RTD can be determined by injecting a conservative tracer at the inlet of the wetland at a known concentration. The tracer concentration is measured through time at various places in the wetland and/or at the outlet. This allows for determination of the flow pattern, actual detention time, volume of stagnant regions, and amount of mixing (dispersion).

The two ideal models for wetland hydraulics are the plug flow model (PF) and the continuous stirred reactor (completely mixed). The plug flow model as mentioned above describes the case where all particles spend and equal amount of time in the reactor and move uniformly from inlet to outlet. In the continuous stirred reactor (CSTR) the particles are uniformly distributed throughout the entire reactor. While neither of these models accurately describes flow in wetlands, modifications of each have been shown to be effective. One variation of the completely mixed reactor is the tank is series model (TIS).

The tank in series model assumes that the wetland is a series of equally sized pieces, which are completely mixed. Figure 4.1 shows the RTDs for the PF, CSTR, and TIS models. The second approach is to use plug flow model modified by dispersion (PFD). Figure 4.2 shows how the dispersion coefficient changes the RTD for the PFD model.

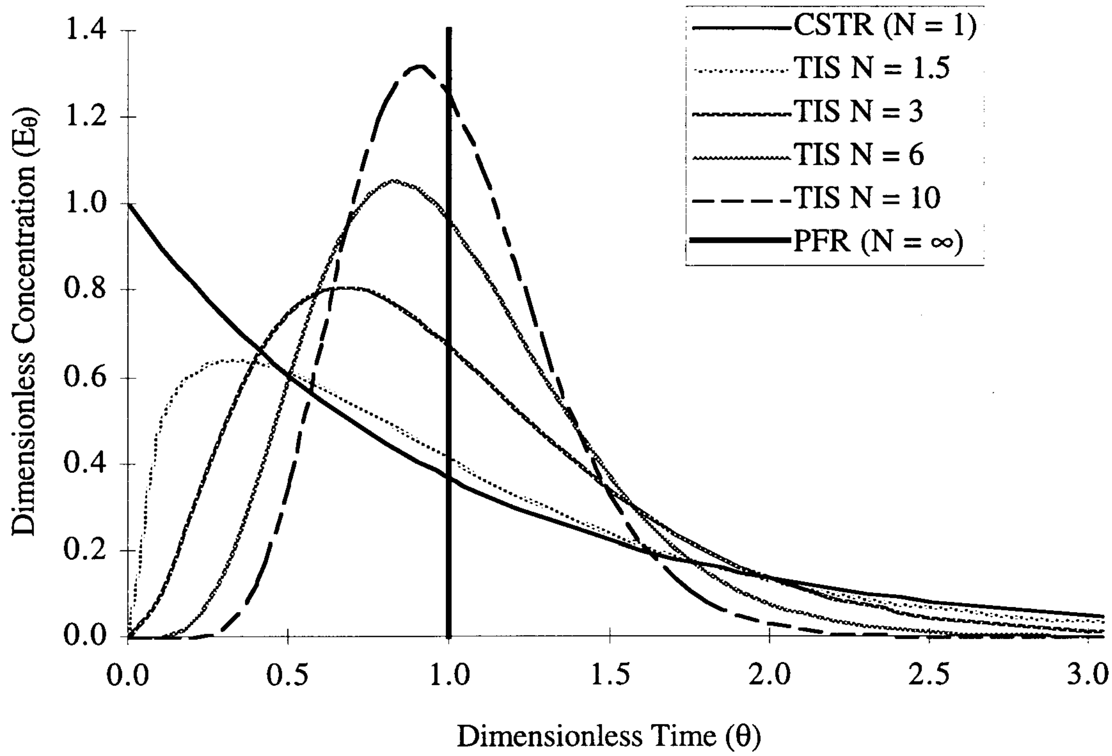

Figure 4.1. Residence time distributions for plug flow reactor (PFR), continuous stirred reactor, and tank in series models (TIS). N = the number of tanks.

The results from tracer studies have shown that constructed wetlands have a flow pattern somewhere between plug flow and completely mixed (Kadlec and Knight, 1996). The TIS and PFD models have both been shown to be effective at predicting flows in some wetlands (Stairs, 1993; Kadlec et al. 1993). Kadlec and Knight (1996) have also shown that a combination of the plug flow model with dispersion and the tank in series model has resulted in better fitting models than either of the models used independently. Kadlec and Knight (1996) suggest that after sufficient tracer studies have been conducted "prototype"

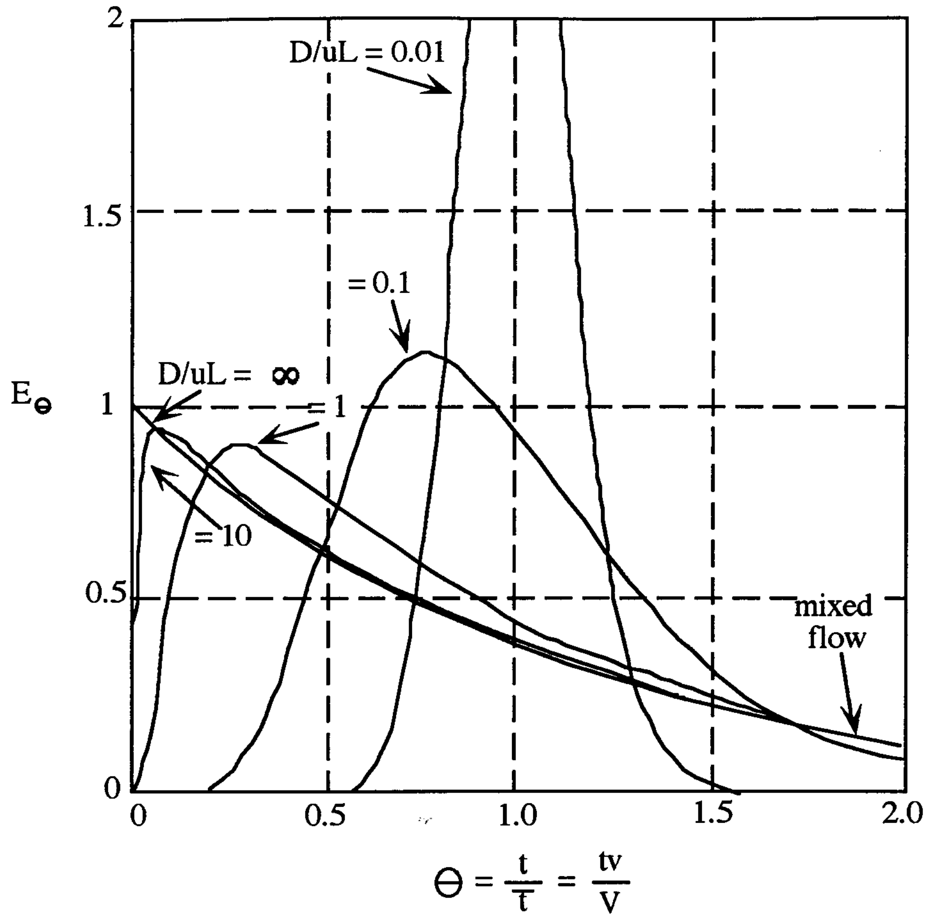

Figure 4.2. Residence time distributions for plug flow modified by dispersion model (PFD) with closed-closed boundary conditions. Adapted from Levenspeil (1993).

wetlands might be used for predicting flow patterns for design of new wetlands. However, few tracer studies have been conducted and until a large database exists it will continue to be necessary to conduct tracer studies at new wetlands. These tracer studies will allow for calculation of accurate design equations, kinetic rates, and treatment efficiencies.

The objectives of this study are to:

1. determine the RTD of the wetland cells at the Oregon State University Dairy Wetland Treatment System (OSUDWTS), and

2. evaluate the fit of the TIS and PFD models.

4.3 Methods

A tracer study was conducted at the OSUDWTS. Residence time distribution theory was used to analyze the data and evaluate the fit of various flow models.

4.3.1 Study Site

The OSUDWTS is located in Corvallis, Oregon and was designed to treat diluted dairy flushwater. The site consists of six parallel wetland cells 28.1 m x 5.9 m x 0.30 m (Ponds 4-9) and a 29.6 m x 10.7 m x 1 m storage pond (Pond 10) (Fig. 4.3). The wetland system was constructed and planted in 1992, began receiving wastewater in October of 1993, and continues to receive wastewater. Treated water is pumped twice daily from pond 10 to a mixing tank where concentrated dairy wastewater is added. The time of day and duration of the pumping of "recycled water" are controlled by a mechanical timer. The concentrated wastewater is loaded from the dairy's pressurized liquid waste handling system using an electric ball valve and electronic timer. The entire volume of the "mixed" wastewater is then loaded to the cells over a period of approximately four hours. The outflows from the wetland ponds drain back into pond 10. The cells are vegetated by a mix of bulrush (*Scirpus acutus* Muhl.), cattails (*Thypha latifolia* L.), and floating grass mats, which were composed of Western mannagrass (*Glyceria occidentallis* (Piper) J.C. Nels.) and water foxtail (*Alopercus geniculatus* L.).

4.3.2 Tracer Study

The tracer used in this study was a solution Rhodamine WT (RhWT), 20% RhWT by weight. RhWT is a fluorescent dye with a minimum detection limit of 0.013 ug/l.

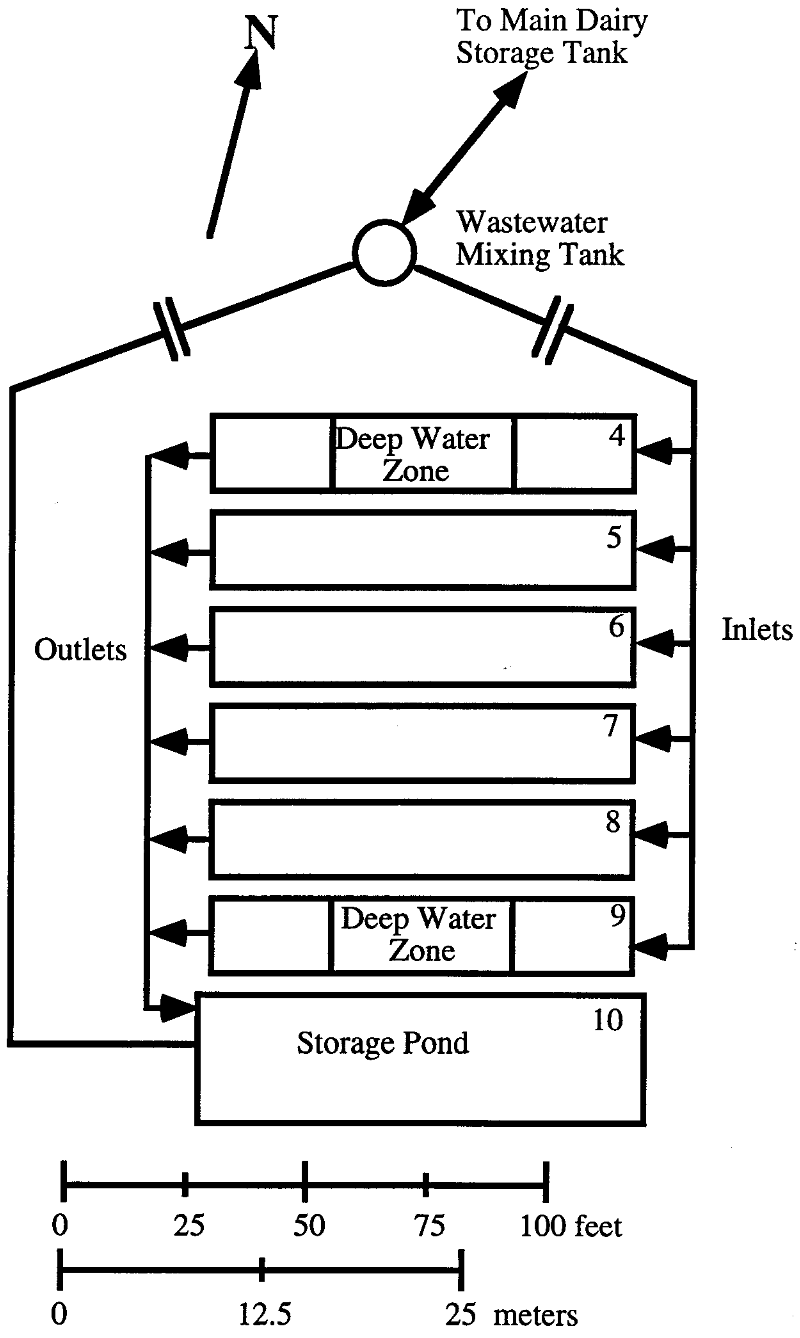

Figure 4.3. Site map of the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Ponds 4-9 are the wetland treatment cells and pond 10 is a storage pond. Lines and arrows indicate pipes and the flow path of wastewater.

Details of selecting a dye are discussed in detail by Denbigh and Turner (1984) and Bowman (1984b). A Turner Fluorometer was used to measure dye concentrations as described by Wilson et al. (1986). The fluorometer was equipped with a GE G4T4/1 UV lamp, which provides a peak excitation of 546 nm. A gel Wratten 61 green filter set between two glass Corning 1-60 gray filters was used as the primary filter. A glass Corning 3-66 and a blue Corning 4-97 filter were used as the secondary filter.

The stock RhWT was used to develop standards (0.5 ug/l to 10.0 ug/l) as described by Wilson et al. (1986). These standards were then used to develop the calibration curves. RhWT fluorescence is extremely temperature sensitive and all samples were stored in a dark incubator at 20°C prior to measurement.

The field study consisted of three separate runs on wetland cells 4, 7, and 8 (Fig. 4.3). These cells were selected because they were the most different from each other. Cell 4 had a deep water section, cell 7 had approximately 50% cattail coverage, and cell 8 had 100% cattail coverage. It was assumed if any differences in hydraulics were going to be observed, then these cells would show the differences. For purposes of this study, freshwater was continuously loaded to the ponds. This allowed for a constant loading rate, decreased the potential sorption of dye to organics found in the wastewater, and also decreased background fluorescence. A dye solution, 10 ml of stock dye mixed with 6 L of water, was added to the inlet as an instantaneous pulse. After the tracer was added, an ISCO automated sampler was used to draw samples at the outlet every 15 minutes. After the peak dye concentration was measured frequency of sampling was decreased to every two hours. Samples were collected for a duration of three times the theoretical detention time for the first run. Sampling was ended early on the second set of runs after the occurrence of a severe rain storm (>4" in 24 hr.). All runs were conducted in November of 1996. The data were entered into a spreadsheet and then normalized. The normalized time (θ) was calculated by dividing the time by the theoretical detention time. The normalized concentration (E_θ) was calculated by dividing the instantaneous dye concentration by the initial dye concentration (Levenspiel, 1993).

4.4 Results

Wetland cells 4, 7, and 8 were tracer tested at the OSUDWTS during November of 1996. All three of the wetland cells had similar residence time distributions.

4.4.1 Theoretical Detention Time

Flow to the ponds was controlled by a flow splitter, which consisted of V notch weirs. Flow into the flow splitter was measured at least daily. Using the flow rates and wetland volume the theoretical detention time (TDT) was calculated. Using the suggested porosity of 0.75, a porosity-corrected detention time was also calculated (Reed et al., 1995). Table 4.1 shows the average flow rates, TDT, and porosity corrected detention times for each pond.

Table 4. 1. Flow rates and corresponding theoretical detention times for Oregon State University Dairy Wetland Treatment System wetland cells.

Wetland Cell	Cell Description	Flow Rate (m^3/d)	Volume of Cell (m^3)	Theoretical Detention Time (days)	Detention Time with porosity correction[1]
4	66% Catail w/ deep section	29.2	63.2	2.16	1.62
7	50% cattail	16.3	50.5	3.10	2.32
8	100% cattail	15.8	50.5	3.20	2.40

[1] porosity assumed to equal 0.75 (Reed et al., 1995)

4.4.2 Residence Time Distributions

The normalized concentrations versus normalized time for all runs are shown in Figure 4.4. The relative shapes of the curves are remarkably similar considering the differences in vegetation in each of the ponds (Table 4.2). All distributions are characterized by a steep rise with an early peak. This is followed by a fairly rapid

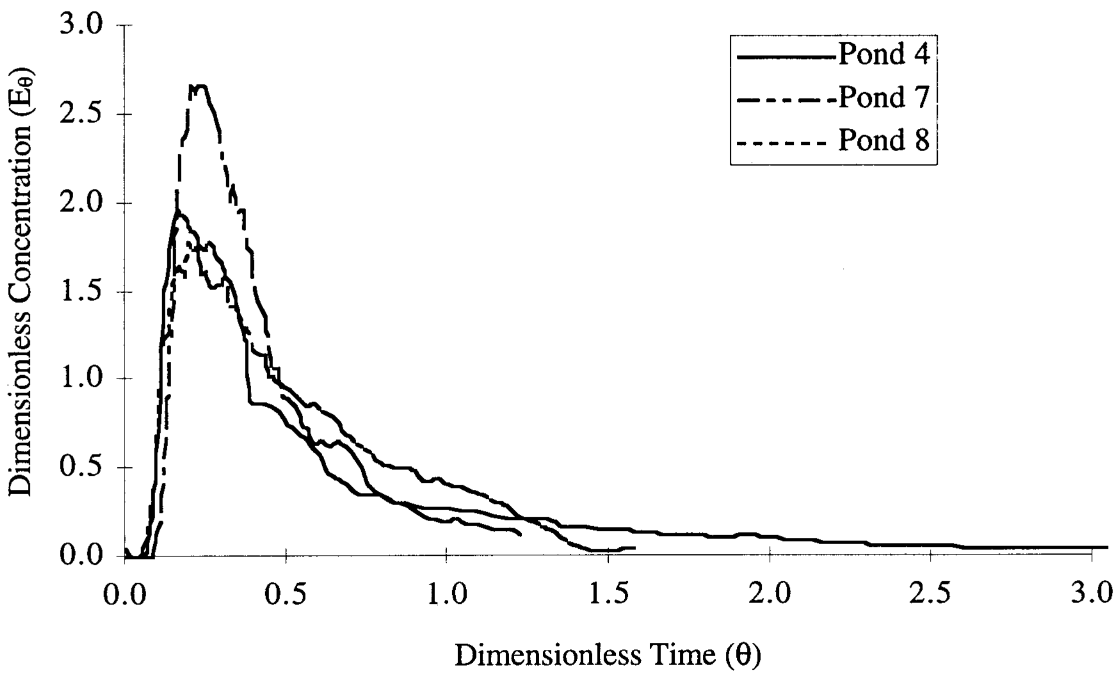

Figure 4.4. Residence time distributions for the Oregon State University Dairy Wetland Treatment System wetland cells, Corvallis, OR.

Table 4.2. Results of normalized concentration vs. time curves for Oregon State University Dairy Wetland Treatment System wetland cells.

Wetland Cell	Cell Description	Time to Peak	Mean	Variance	Dye Recovery
4	66% Catail w/ deep section	0.16	0.60	0.28	74%
7	50% cattail	0.23	0.56	0.28	61%
8	100% cattail	0.22	0.56	0.16	41%

exponential decrease followed by a long gradual falling tail. The gradual falling tail may indicate several things:

1. a large amount of dispersion, and/or
2. the existence of dead space, and/or
3. sorption-desorption of dye.

Other researchers have also noted long tails when using RhWT dye (Eberdorfer, 1993; Stairs, 1993; Kadlec and Knight, 1996). Ideally when conducting tracer studies, 100% of the dye is recovered within three detention times. If the data show a long tail then the conclusions drawn from the RTD may be erroneous (Levenspiel, 1972). Unfortunately, fluorescent dyes are notorious for sorption and desorption and often give long tails. Kadlec and Knight (1996) suggest that in the case of long tails an exponential decreasing function can be fit to the tail. This function can be extrapolated from the data past the second inflection point of the residence time distribution (Kadlec and Knight, 1996). Such a function was fit to each of the runs (Fig. 4.5). The time to peak, mean detention time, variance, and dye recovery for each wetland cell are summarized in Table 4.2.

The time to peak for pond 4 is slightly quicker than pond 7 and 8. However, the means for each cell are almost the same. The variance for pond 8 is less than the variance of ponds 4 and 7. The dye recoveries were low for all ponds and extremely low for pond 8. This recovery, however, is comparable with the recovery reported by Stairs (1993) and Eberdorfer (1993). As mentioned above, failure to recover 100% of the dye may bring into the question of the validity of the RTD. Nevertheless it was assumed that the RTD accurately represented the general hydraulic characteristics.

In addition to the mean DT and time to peak, the RTD can also be used to calculate the effective volume and volume of dead space. The effective volume of a reactor

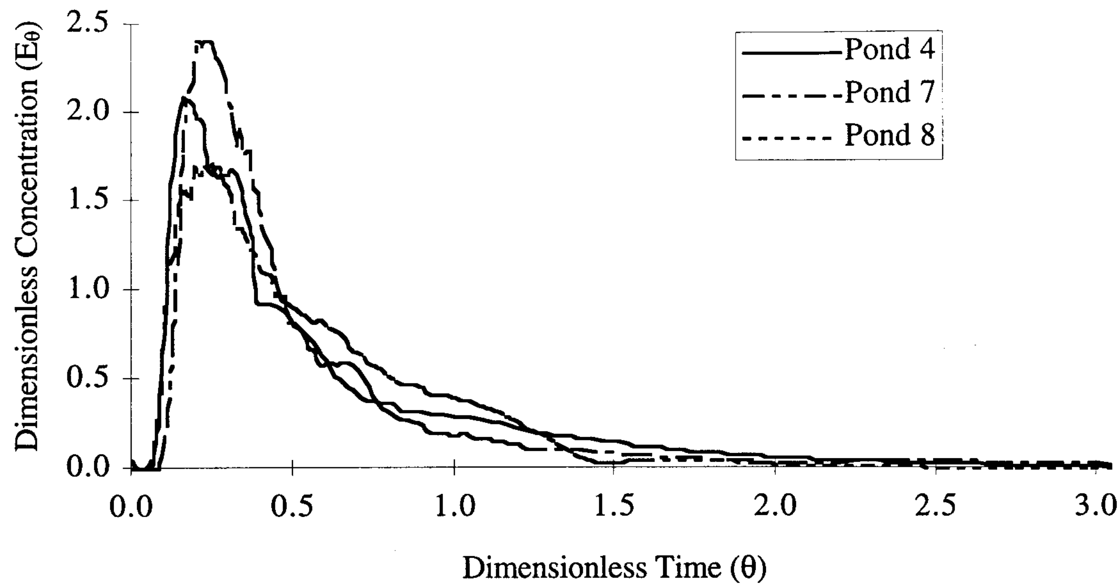

Figure 4.5. Residence time distributions for the Oregon State University Dairy Wetland Treatment System wetland cells, Corvallis, OR, with fit tails.

(wetland) is calculated by determining the size of reactor that would be needed to produce the mean residence time found from the RTD:

$$V_e = DT_{mean} \cdot Q \qquad (4\text{-}1)$$

where, V_e = effective volume (m^3),

DT_{mean} = mean detention time (d), and

Q = water loading rate (m^3 /d).

The volume of "dead space" is the actual volume of the wetland minus the effective volume. The effective porosity equals the effective volume divided by the actual volume. Effective porosities for constructed wetlands typically range from 0.75 to 0.90 (Reed et al., 1988; Kadlec and Knight, 1996). The porosity takes into account two phenomena, the volume occupied by plants and litter and the volume of the reactor not actively involved in the flow ("dead space"). Obviously, the volume occupied by litter and vegetation depends on the density, productivity, and type of vegetation. The amount of dead space in a wetland is dependent on the flow pattern. The flow pattern is influenced by the location of the inlet and outlet, use of flow distribution devices, length to width ratio, depth, vegetation type and density, shape of wetland, and microtopographic features. The effective porosity for ponds 4, 7, and 8 were 0.60, 0.56, and 0.56, respectively, which is considerably lower than the average reported values (0.75 to 0.9) (Table 4.3).

4.4.3 Tank in Series Model

Flow models can give more insight into the hydraulics of wetlands than just the mean DT, dead space, and effective volume. Conceptually, the tank in series model (TIS) is a linear series of completely mixed reactors of equal size. The number of tanks can be calculated from the dimensionless variance reported in Table 4.2 (Levenspiel, 1972):

$$\sigma^2 = 1/N \qquad (4\text{-}2)$$

where, σ^2 = dimensionless variance, and

N = number of tanks.

Most wetlands that have been tracer tested yielded residence time distributions that suggested from two to eight tanks. However, most are between two and five with and average of three (Kadlec and Knight, 1996). The number of tanks for this study was 3.6, 3.6, and 6.3 for ponds 4, 7, and 8, respectively. Figures 4.6, 4.7, and 4.8 show the fit of these models to the data. It is apparent from these figures that the TIS model does not do a good job of predicting the shape of the RTD. This is largely due to the fact that there is a large amount of dead space in the wetland and/or the dye is being absorbed and released (Levenspiel, 1993). If the effective volume is used (i.e. the dead space is ignored) and the tail is ignored a slightly better fit is achieved (Fig. 4.9, 4.10, 4.11). The number of tanks

Table 4.3. Effective volume, dead space, and porosity of Oregon State University Dairy Wetland Treatment System wetland cells.

Wetland Cell	Theoretical Detention Time (d)	Mean Detention Time (d)	Flow Rate (m^3/d)	Volume of Cell (m^3)	Effective Volume (m^3)	Dead Volume (%)	Effective Porosity
4	2.16	1.30	29.2	63.2	37.9	40%	0.60
7	3.10	1.73	16.3	50.5	28.3	44%	0.56
8	3.20	1.79	15.8	50.5	28.3	44%	0.56

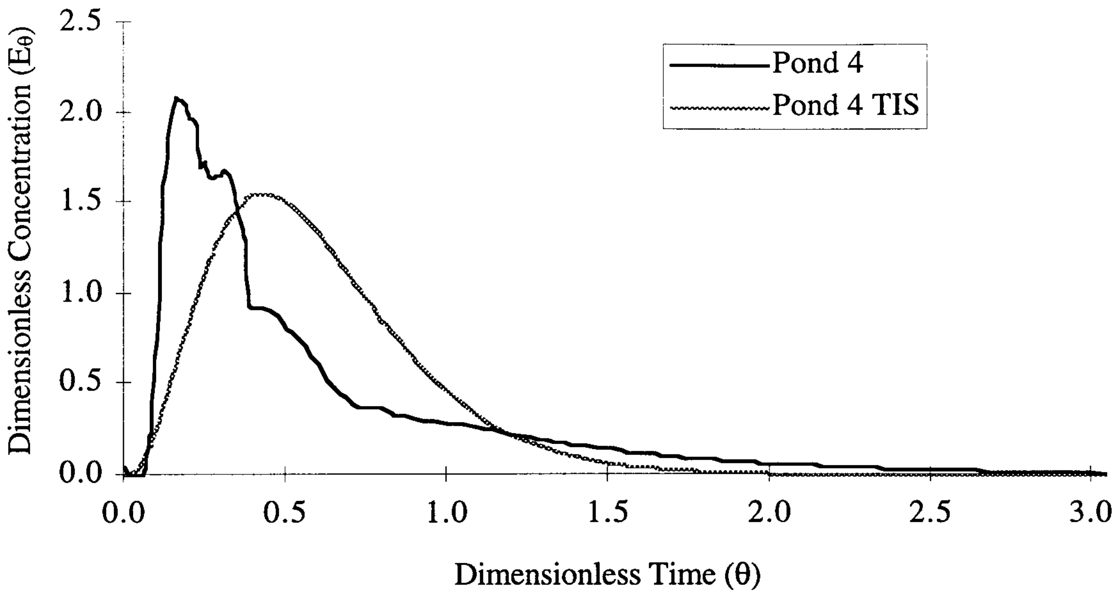

Figure 4.6. Fit of the tank in series model (TIS) to the residence time distribution of Pond 4 at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR (N= 3.6; R^2 = 0.43).

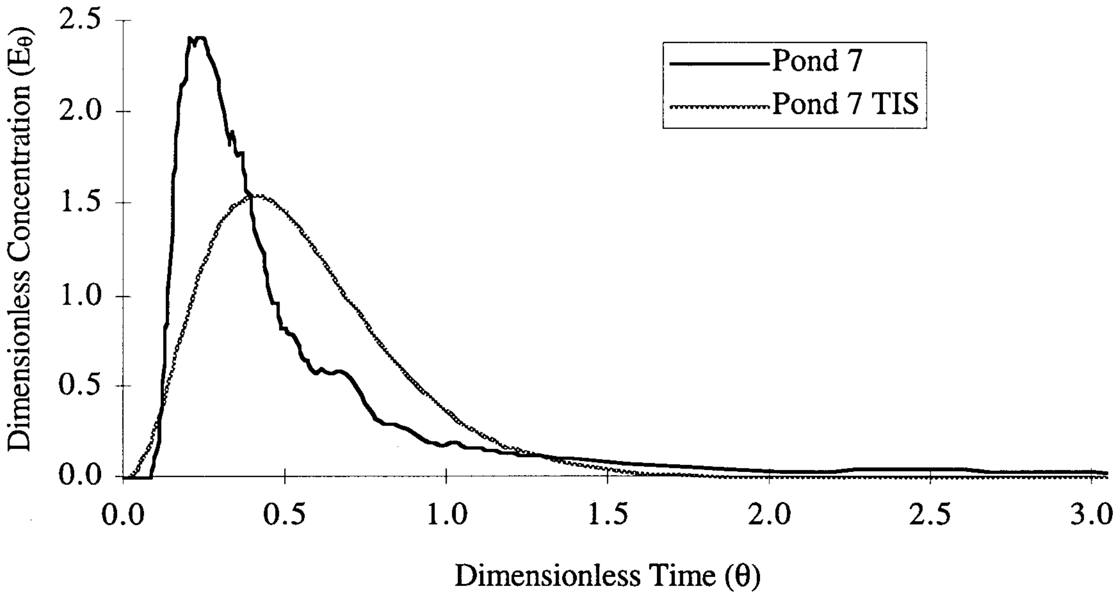

Figure 4.7. Fit of the tank in series model (TIS) to the residence time distribution of Pond 7 at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR (N = 3.6; R^2 = 0.50).

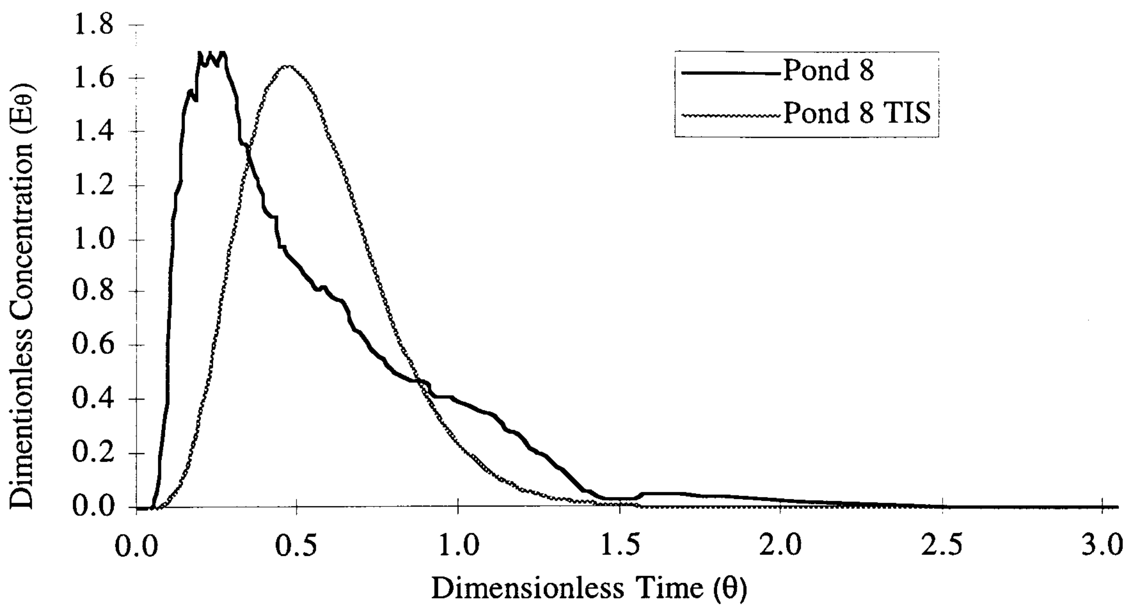

Figure 4.8. Fit of the tank in series model (TIS) to the residence time distribution of Pond 8 at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR ($N = 6.3$; $R^2 = 0.32$).

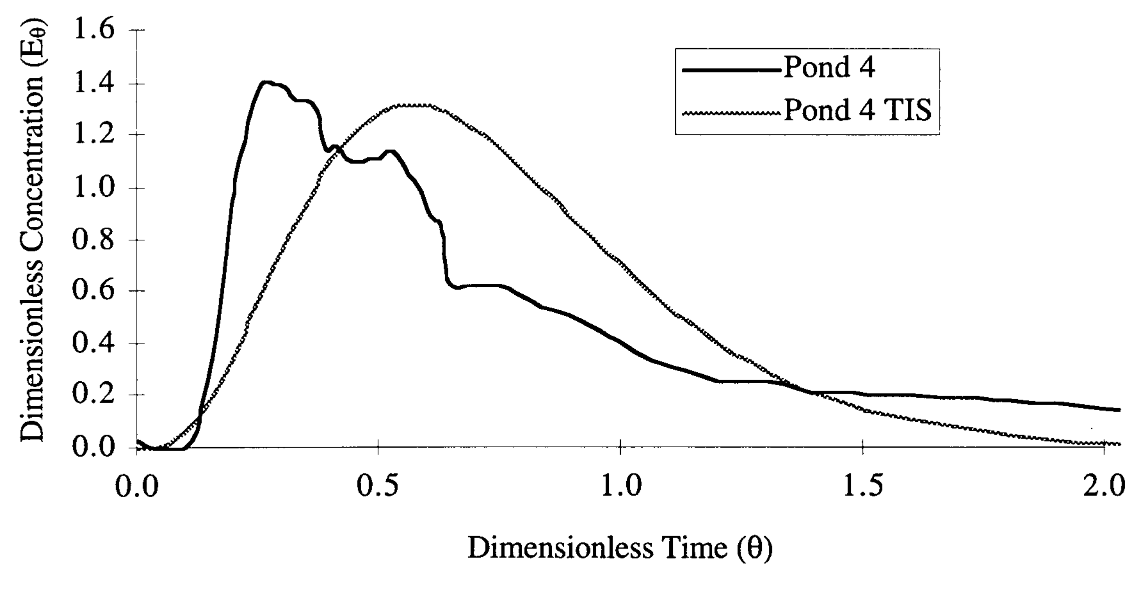

Figure 4.9. Fit of the tank in series model (TIS) to the residence time distribution of Pond 4 using the effective volume of the wetland cell at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR ($N = 4.3$; $R^2 = 0.52$).

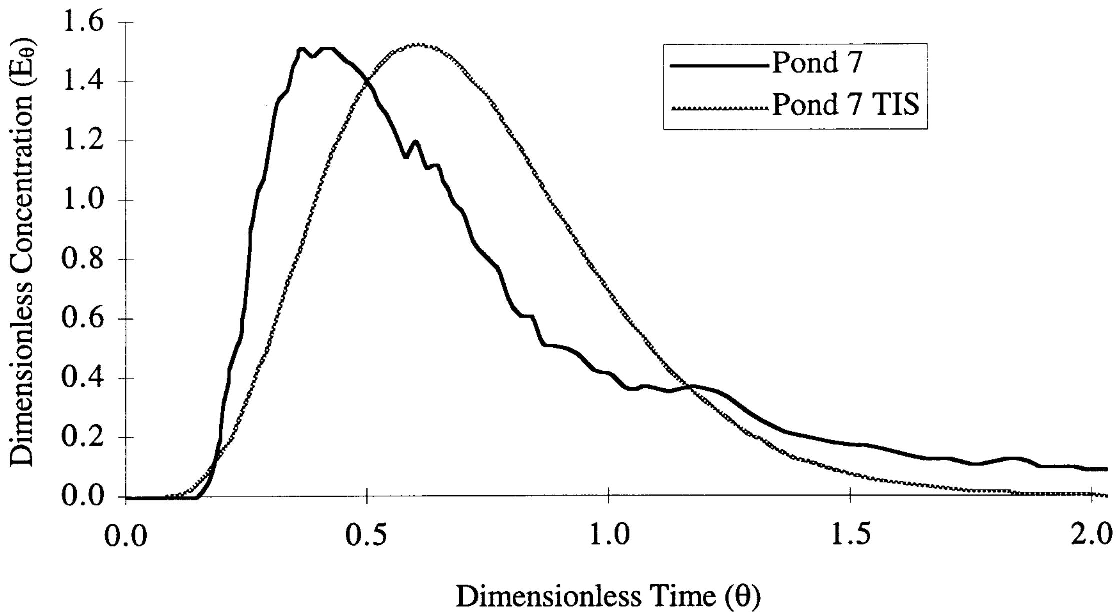

Figure 4.10. Fit of the tank in series model (TIS) to the residence time distribution of Pond 7 using the effective volume of thewetland cell at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR (N = 6.2; R^2 = 0.65).

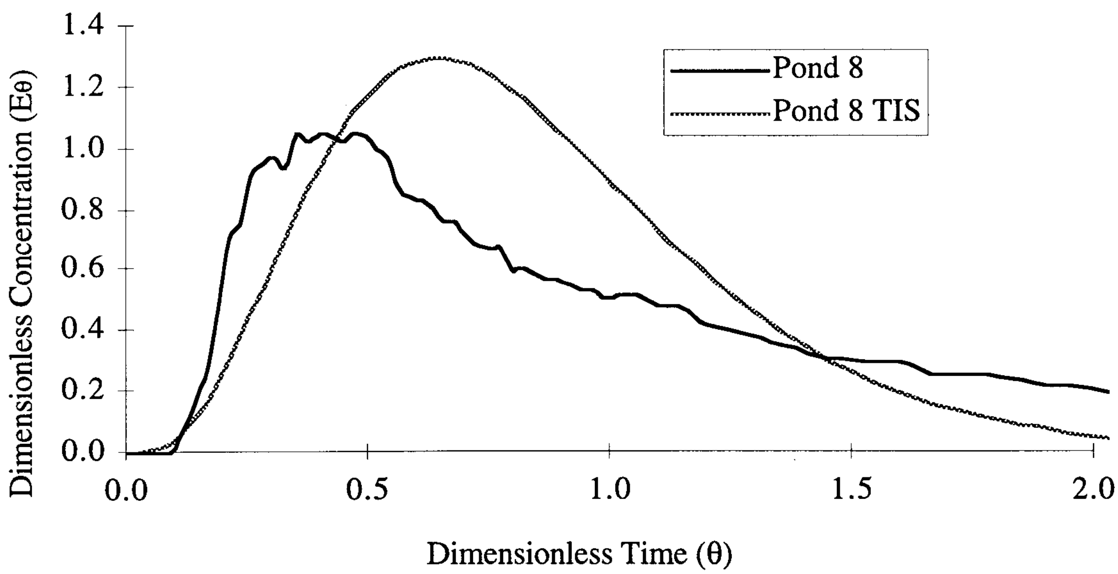

Figure 4.11. Fit of the tank in series model (TIS) to the residence time distribution of Pond 8 using the effective volume of the wetland cell at the Oregon State University Dairy Wetland Treatment System, Corvallis, OR (N = 4.3; R^2 = 0.65).

becomes 4.3, 6.2, and 4.3 and the R^2 = 0.52, 0.65, and 0.65 for ponds 4, 7, and 8 respectively.

4.4.4 Plug Flow Modified by Dispersion Model

The plug flow modified by dispersion model (PFD) has also been used to model wetland hydraulics. It is important that the appropriate boundary conditions be chosen when using this model. Kadlec and Knight (1996) state that the open-open case has been repeatedly misused for modeling wetland hydraulics (Bavor et al., 1988; Stairs, 1993). For the case of wetlands, the closed-closed boundary conditions must be used (Fogler, 1992). This case states that no tracer can diffuse back into the outlet nor back up the outlet of the wetland. Unfortunately, no closed-form solutions exist for the closed-closed case but numerical solutions do exist to construct the RTD (Yagi and Miyauchi, 1953). Fortunately, the mean and variance from the RTD can be used to calculate the wetland dispersion number directly (Levenspiel, 1993):

$$\sigma^2 = 2\frac{D}{(uL)} - 2\frac{D}{(uL)}(1 - e^{(uL)/D}) \qquad (4\text{-}3)$$

where, σ^2 = dimensionless variance,

D = dispersion constant (m^2/d),

u = velocity in x direction (m/d) = *L/DT,*

L = distance from inlet to outlet (m), and

DT = detention time (d).

The dimensionless dispersion number $\left(\frac{D}{(uL)}\right)$ is often used to describe reactors and allows comparison of different sized reactors. Typical values of $\left(\frac{D}{(uL)}\right)$ for wetlands range from

0.07 to 0.33 (Kadlec and Knight, 1996). The values of $\left(\frac{D}{(uL)}\right)$ for pond 4, 7, and 8 using the actual volume of the ponds were 0.17, 0.17, and 0.09, respectively. These all fall within the typical range reported for constructed wetlands.

The one dimensional plug flow modified by dispersion for equation states (Kadlec and Knight, 1996):

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial x^2} - \frac{\partial (uC)}{\partial x} \qquad (4\text{-}4)$$

where, u = velocity (m/d),

D = dispersion constant (m^2/d),

x = distance from inlet toward outlet (m),

t = time (d), and

C = concentration of tracer (g/m^3).

The plug flow model modified by dispersion gives a similar shaped curve to the TIS models (see Fig. 4.1 and 4.2). The plug flow model modified by dispersion fails to do a very good job of describing the flow pattern at OSUDWTS for the same reasons discussed in the previous section.

4.5 Discussion

The shapes of the RTD for all three ponds were remarkably similar considering that each of the ponds has a different percentage of plant cover and pond 4 has a deep water section. This may indicate that the length to width ratio and design of inlet and outlet structures played a more important role than the vegetation in determining the flow characteristics. The outlet structure consist of a single point discharge through a 4" PVC pipe. The inlet structure is also known to produce a less than ideal distribution. This configuration may cause the corners of the wetland cell to be isolated from the main flow path causing the large amount of dead space. This is supported by Stairs (1993) findings that when an outlet was modified from a pipe that collected flow across the entire width of the pond to a point discharge that the detention time was greatly decreased.

The time to peak was also very quick for all ponds, indicating that preferential flow ("short circuiting") is occurring. The mean detention times (0.56-0.60) were near identical for all ponds. This detention time is much shorter than theoretical and again indicates a high volume of dead space and short circuiting. The long tails also indicate the presence of dead space and a slow exchange of dye between the active volume and stagnant regions.

If the theoretical detention time was used to calculate treatment performance, it would over predict the expected treatment. Removal of biochemical oxygen demand (BOD) can be used to show how use of the theoretical detention time would result in erroneous errors. Assume BOD removal can be modeled using a first order irreversible reaction:

$$C_{out} = C_{in}^{-kt} \qquad (4\text{-}5)$$

where, C_{out} = BOD concentration leaving wetland (mg/l),

C_{in} = BOD concentration entering wetland (mg/l),

k = BOD kinetic coefficient (1/d), and

t = detention time (d).

Using a conservative estimate of $k = 0.2$ 1/d, $C_{in} = 100$ mg/l, and the theoretical and mean detention times (Table 4.3), the percent BOD removal can be calculated (Table 4.4). The removal based on the theoretical detention time was an average of 15% higher than the removal based on the mean detention time. This is a large difference and shows the importance of taking into account the hydraulics of a wetland.

Table 4.4. BOD removal based on theoretical and mean detention times (DT) of the wetland cells at the Oregon State University Dairy Wetland Treatment System. k = 0.2 1/d. C_{in} = 100 mg/l.

			Using Theoretical DT		Using Mean DT		
Wetland Cell	Theoretical DT (d)	Mean DT (d)	C_{out} (mg/l)	% Removal	C_{out} (mg/l)	% Removal	Difference (%)
4	2.16	1.30	64.9	35%	77.1	23%	-12%
7	3.10	1.73	53.8	46%	70.7	29%	-17%
8	3.20	1.79	52.7	47%	69.9	30%	-17%
						Average =	-15%

4.6 Conclusions

Currently, wetlands are designed based on the plug flow assumption and the theoretical retention time, which has been shown to be incorrect. In addition, kinetic coefficients are largely based on data collected at wetlands that have not been tracer tested. This may explain a good portion of the variability seen in kinetic coefficients and treatment results.

Use of fluorescent dyes as tracers in wetlands is not appropriate. Sorption-desorption and photodegradation appear to be high and causes low dye recovery. Low dye recovery brings into question the validity of the RTD. More conservative dyes such as bromide may be more appropriate. Inorganic and organic anions (Bowman, 1984a; Bowman, 1984b; Bowman and Rice, 1986) and fluorobenzoate tracers (Pearson et al. 1992; Bowman and Gibbens, 1992) were shown to be very good tracers in soil environments and may also be very useful in wetlands.

The wetland cells at the OSUDWTS have very short mean detention times and a large amount of "dead space", indicating that short circuiting is occurring. This problem could possibly be corrected by installing better distribution pipes at the inlet and outlet. The TIS and PFD models are not effective at modeling the flow pattern in the wetland cells at the OSUDWTS. The removal based on the theoretical detention time was an average of 15% higher than the removal based on the mean detention time.

This study provides additional evidence that hydraulics must be taken into account when evaluating constructed wetland systems. If the science of wetland design is to advance then kinetic coefficients should also be developed taking into account the hydraulics.

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5. Treatment of Dairy Wastewater Using a Constructed Wetland

5.1 Abstract

The use of a constructed wetland system for the treatment of concentrated dairy wastewater was evaluated in Corvallis, OR. Six parallel wetland cells, each 28.1 m by 5.9 m by 0.30 m, were constructed in 1992 and they began receiving wastewaters in the fall of 1993. Data were collected on 39 sampling dates between October 1993 to December 1996. Average loading of BOD and TKN was 188 and 55 kg/ha-d. Average reductions for COD, BOD, TS, TSS, TP, TKN, NH_3 and fecal coliforms were 45, 52, 27, 55, 42, 41, 37 and 80%, respectively.

Rate constants for volumetric and areal first-order plug flow models were determined for each wetland cell and waste parameter. Both volumetric and areal models resulted in similar fits and explained similar proportions of the variability ($\overline{R}^2$ = 0.65 and 0.67, respectively). The models did not fit the TSS data and resulted in a $\overline{R}^2$ of 0.03 and 0.01 for the volumetric and areal models, respectively. The volumetric rate constants were found to not be temperature dependent ($\overline{\theta}$ = 1.01; range 0.99 to 1.02). The areal rate constants were slightly temperature dependent ($\overline{\theta}$ = 1.04; range 1.01 to 1.12).

This study showed that constructed wetlands can have high removal rates for many common wastewater constituents. However, the reductions were slightly less than those reported for constructed wetlands treating domestic wastewater. It appears that oxygen limitations, resulting from the high strength livestock wastewater, may be causing the lower removal rates.

Keywords: design models; first-order plug flow models

5.2 Introduction

Use of surface flow constructed wetlands for treatment of livestock wastewater is increasing. At least 68 wetland systems have been built in North America for treatment of livestock wastewaters (Knight et al., 1996 draft). Preliminary results indicate potentially high removal of biochemical oxygen demand, total suspended solids, total nitrogen, total phosphorus, and fecal coliforms (Knight et al., 1996 draft). However, performance data are highly variable and only simple empirical equations are available to predict treatment. Design equations have been developed for surface flow wetlands treating domestic wastewater, which are based on first-order models (Kadlec and Knight, 1996; Reed et al., 1995). However, there is no general consensus among wetland scientists whether areal or volumetric based first-order models are most appropriate (Reed, 1995 draft). Efforts are underway (supported by U.S. EPA.) to produce a manual for "Free water surface constructed wetlands for wastewater treatment: a technology assessment" (Kadlec and Knight, 1996 draft). This assessment, however, is being developed for wetlands treating domestic wastewaters. It is not known if the design equations and rate constants developed for these low concentration wastewaters will be valid for high strength livestock waste.

If constructed wetlands are going to be accepted as a treatment technology for livestock wastewater, reliable design and removal equations are required. Of the 68 constructed wetland sites in the Livestock Wastewater Treatment Wetland Database, only five have sufficient data to predict first-order rate constants (Knight et al., 1996 draft).

It is the objectives of this study to:

1. determine the treatment performance of the Oregon State University Dairy Wetland Treatment System (OSUDWTS) for chemical oxygen demand (COD), five day biochemical oxygen demand (BOD_5), total solids (TS), total suspended solids (TSS), total phosphorus (TP), orthophosphate (PO_4-P), total Kjeldahl nitrogen (TKN), ammonia (NH_3), nitrate (NO_3^-), and fecal coliforms;

2. develop volumetric and areal based first-order rate constants for each wastewater parameter; and
3. evaluate the fit of the volumetric and areal models.

5.3 Background

In the U.S., surface water impairment occurs in 332,000 km of rivers, 215,000 ha of lakes, and 1.5 x 106 ha of estuaries, despite national efforts to reduce point source pollution (U.S. EPA, 1990). The U.S. EPA (1986) estimated that two-thirds of these waters are impaired as a result of nonpoint source pollution, which contributes over 65% of the total pollutant load to U.S. inland waters (U.S. EPA, 1989). Nonpoint sources include stormwater runoff from suburban and urban areas, diffuse runoff agricultural and forestry lands, and concentrated runoff from mining areas and concentrated agricultural activities, such as feedlots. It is estimated that agricultural runoff is the single greatest source of nonpoint-source pollution (U.S. EPA, 1990).

These findings have led to increased emphasis on controlling nonpoint source pollution from agricultural lands. In particular, confined animal feeding operations (CAFO), have been targeted as areas that need improvement. In general, CAFOs have some sort of existing treatment system, such as anaerobic lagoons and/or spray irrigation systems for land application. However, in most cases these systems are inadequate for treating 100% of the wastewater to sufficient levels. Constructed wetlands have been successfully used to treat domestic and industrial wastewater since the 1970's and are currently being evaluated for treatment of nonpoint source pollution and concentrated animal waste. Numerous conferences have been held and several books written about constructed wetlands treating domestic and industrial wastewaters (Kadlec and Knight, 1996; Reed et al., 1995) Most of this information is applicable to constructed wetlands treating livestock waste but there are currently no tested design equations for these systems.

In 1994, Purdue University sponsored the first national workshop dedicated to "Constructed Wetlands for Animal Waste Management" (DuBowy and Reaves, 1994). Eighteen papers were presented, which were mostly about research currently underway or in the development stages. In 1995, the Gulf of Mexico Program (GMP), an inter-agency group sponsoring research to enhance the environmental and economic viability of the Gulf of Mexico, sponsored the Alabama Soil and Water Conservation Committee and the National Council of the Pulp and Paper Industry for Air and Stream Improvement (NCSAI) efforts to review the use of constructed wetlands for animal wastewater treatment (Knight et al., 1996 draft). The goals of this project were to develop (Knight et al., 1996 draft):

1. a literature review of constructed wetlands for treating concentrated livestock wastewater (Payne, 1996 draft);
2. a database (the Livestock Wastewater Treatment Wetland Database) of design and operational data for these systems (Knight et al., 1996 draft); and
3. a public outreach brochure to help agricultural managers consider the advantages and disadvantages of these systems.

The literature review states that constructed wetlands may be used as part of a animal waste management system for the purposes of (Payne, 1996 draft):

1. nutrient matching - excess nutrients from manure that can not be land applied can be treated with a constructed wetland;
2. pollutant reduction - where applicable constructed wetlands can be used to meet discharge standards;
3. odor control - wetland effluent is relatively odorless compared with traditional treatment technologies; and
4. labor reduction - use of a constructed wetland may decrease waste handling and the amount of time spent land applying wastewater due to the decreased volume of wastewater.

The literature review also states that the use of constructed wetlands for treatment of livestock waste is particularly attractive for several reasons (Payne, 1996 draft):

1. wastewater is typically isolated and undergoing pretreatment, which makes delivery to the wetland system easy;
2. land is generally available for construction of the wetland; and
3. construction and maintenance costs of constructed wetlands are often lower than traditional treatment technologies.

The Livestock Wastewater Treatment Database (LWDB) that was developed includes 68 sites with 138 pilot and full scale wetland systems treating livestock wastewater (Knight et al., 1996 draft). Preliminary analysis of the database indicates that constructed wetlands can achieve high removal rates of biochemical oxygen demand, total suspended solids, total nitrogen, total phosphorus, and fecal coliforms (Knight et al., 1996 draft). The second national workshop for "Constructed Wetlands for Animal Waste Management" was held in May 1996 and over 30 papers were presented. Papers presented included several pilot studies, future research needs, and preliminary findings from the GMP research project.

At the workshop, several challenges were identified that must be addressed before constructed wetlands become an accepted technology for treating livestock waste. These include:

1. the ability to predict treatment and outlet concentrations;
2. design equations and criteria for sizing new wetlands; and
3. a commitment by livestock producers to manage these systems and regulators to recognize constructed wetlands as part of a producer's waste management plan.

5.4 Design Equations

There is a general consensus among designers of constructed wetlands that a first-order plug flow model is adequate for describing treatment in surface flow wetlands given the current data available (Reed, 1995 draft). The first-order plug flow model states:

$$\frac{C_{out}}{C_{in}} = e^{-k \cdot t} \qquad (5\text{-}1)$$

where, C_{out} = effluent concentration (mg/l),

C_{in} = influent concentration (mg/l),

k = rate constant (appropriate units), and

t = time (units consistent with k).

This model is used to predict treatment for many of the common wastewater constituents such as COD, BOD, TS, TN, TP and fecal coliforms. Two common methodologies for the use of the first-order plug flow model exist. The first is based on volumetric rate constants (Reed et al., 1995) and the second on areal rate constants (Kadlec and Knight, 1996). There is disagreement among wetland designers on the appropriateness of each of these models. A summary of the differences and similarities between the two models is presented in Table 5.1. The models will be discussed in more detail below.

5.4.1 Volumetric Based First-order Plug Flow Model

The equation for the volumetric based first-order plug flow model is (Reed at al., 1995):

$$C_{out} = C_{in} \cdot e^{\left[\frac{-(k_t \cdot A \cdot d \cdot n)}{Q}\right]} \qquad (5\text{-}2)$$

where, C_{out} = effluent concentration (mg/l),

Table 5.1. Summary of differences and similarities between the volumetric (Reed et al., 1995) and areal (Kadlec and Knight, 1996) based first-order plug flow models.

Parameter	Volumetric Based	Areal Based
C_{in} & C_{out}	based on individual samples (mg/l)	time average of samples (minimum of monthly); takes into account variation in waste strength, and dilution and concentration effects; must be several times theoretical retention time (mg/l)
k_t	temperature dependent for most waste constituents (d^{-1})	temperature dependent for most waste constituents (m/yr)
Temperature Dependence	described by Arrenhius equation (Eq. 5-3)	described by Arrenhius equation (Eq. 5-3)
C^*	none; specified as a lower boundary condition	incorporated into model (mg/l)
Q	average of inlet and outlet flows; takes into account gains and losses due to rain, ET, and infiltration (m^3/d)	average of long term loadings (gains and losses balance); use annual loading rate (m/yr)
A	surface area of wetland (m^2)	surface area of wetland (m^2)
d	average depth of wetland; as depth increases detention time increases and therefore, treatment increases	depth does not effect treatment
n	0.65-0.75 (decimal fraction)	not used
Mechanism for Treatment	surface attached microorganisms on plant material and soil primarily responsible	treatment dependent only on surface area of soil-water interface

C_{in} = influent concentration (mg/l),

A = surface area of constructed wetland (m^2),

k_t = temperature dependent first-order rate constant (d^{-1}),

d = average wetland depth (m),

n = wetland porosity (% as a decimal), and

Q = average flow rate (m^3/d).

C_{out} (mg/l) and C_{in} (mg/l) are the concentrations based on a single sampling event. The first-order rate constant k_t is reported to be temperature dependent for most water quality parameters (Reed, 1995 draft). The temperature dependence can be described by the Arrhenius equation (Kadlec and Knight, 1996):

$$k_t = k_{20} \cdot \theta^{(T-20)} \qquad (5\text{-}3)$$

where, $k_{20} = k_t$ (d^{-1}) at 20°C,

$k_t = k$ (d^{-1}) at T (°C),

θ = theta value (dimensionless), and

T = water temperature (°C).

If just the exponential term, $\left[\frac{-\left(k_t \cdot A \cdot d \cdot n\right)}{Q}\right]$, is examined it becomes apparent that the term is simply the rate constant times the mean detention time. Mean detention time is:

$$t = \frac{V_w}{Q} = \frac{A \cdot d \cdot n}{Q} \qquad (5\text{-}4)$$

where, V_w = wetland volume (m^3).

The porosity term (n) is defined as the volume available for water to flow (U.S. EPA., 1988). This porosity term was developed to account for the space occupied by wetland vegetation and litter. Reed et al. (1995) suggest using a porosity of 0.65 to 0.75 for constructed wetlands. This range has been disputed, however, and a wide range of porosities are cited by Payne (1996 draft): cattails, bulrush, reeds, wood grass and rushes occupied 10%, 14%, 2%, 6% and 5% of the wetland volume respectively. Rogers et al. (1995) reported fill rates of 10% for *Sagittaria lancifolia*, and 7% for *Phragmites australis*. It would appear that Reeds' porosity values are too high. However, considering the definition of n "the volume available for water to flow," not only must the space occupied by plants be considered but also the "dead space" (i.e. the volume not actively involved in flow). It has been shown using tracer studies that constructed wetlands have effective porosities ranging from 0.6 to 0.8 (see Chapter 4; Stairs, 1993; Reed et al., 1995; Kadlec and Knight, 1996). Inclusion of the porosity term in Eq. 5-4 takes into account not only the volume occupied by plant material but also the "dead space."

Depth (d) is also included in the exponential term and is one of the critical differences between the volumetric and areal first-order plug flow models. The volumetric model is based on the premise that if depth is increased detention time will increase and therefore treatment will increase (Reed et al., 1995). Underlying this premise is the assumption that plant stems and plant litter provide surfaces for attached growth organisms, which are primarily responsible for waste degradation and removal. Therefore, as depth increases both the time for treatment increases and the microbial biomass responsible for treatment increases (Reed, 1995 draft). Conversely, the areal based model assumes that most of the microbial activity and treatment occurs at the soil-water interface, and increased depth and detention time provide no additional treatment (Reed, 1995 draft).

The flow rate (Q) is based on the average of the inlet and outlet flow rates and takes into account the effect of water gains and losses due to infiltration, precipitation , and evapotranspiration.

The three unknowns in Eq. 5-2 (k_{20}, θ and n) must be found before the equation can be used for design purposes. If temperature data, inlet and outlet concentrations, and inlet and outlet flows are available for a constructed wetland then k_{20}, θ, and n can be solved for simultaneously by using nonlinear regression while minimizing the sum of squared errors between predicted and actual outlet concentrations.

Equation 5-2 can also be rearranged for design of constructed wetlands:

$$A = \left(\frac{Q}{k_t \cdot d \cdot n}\right) \cdot \ln\left(\frac{C_w}{C_{out}}\right) \qquad (5\text{-}5)$$

Inputting the loading rate, inlet and desired outlet concentrations, porosity, depth, and appropriate rate constant into Eq. 5-5, one can find the surface area of the wetland needed to give the desired treatment. After design of the wetland, if C_{out} is not meeting expected levels, depth can be increased to increase performance.

5.4.2 Areal Based First-order Plug Flow Model

The areal based first-order plug flow model is (Kadlec and Knight, 1996):

$$C_o = C^* + \left(C_{in} - C^*\right) \cdot e^{\left[\frac{-(k_t \cdot A)}{Q}\right]} \qquad (5\text{-}6)$$

where, C_o = time average effluent concentration (mg/l),

C^* = background (residual) concentration (mg/l),

C_{in} = time average influent concentration (mg/l),

A = surface area of constructed wetland (m^2),

k_t = temperature dependent first-order rate constant (m/yr), and

Q = annual flow rate (m^3/yr).

The inlet and outlet concentrations are based on the time average concentration which is defined as (Kadlec and Knight, 1996):

$$\bar{C} = \frac{1}{t_m}\int_0^{t_m} C \cdot dt \qquad (5\text{-}7)$$

where, $\bar{C}$ = time average concentration (mg/l),

C = concentration (g/m^3),

t_m = time period for averaging (d), and

d = average wetland depth (m).

Using the average concentration under steady flow conditions eliminates the need to describe short term fluctuations due to the variability in the wastewater strength and flows (Kadlec and Knight, 1996). Kadlec and Knight (1996) state that the time averaging period must be:

1. several times the theoretical detention time in order to avoid transit time delays; and
2. long enough that water gains and losses balance.

Generally, monthly, quarterly, or annual time periods are used (Kadlec and Knight, 1996).

C^* is the background constituent concentration (mg/l) and is the expected concentration one would find in a natural wetland or constructed wetland not being loaded with wastewater. It represents the lowest concentration that can be achieved for a given waste constituent.

A is the area of the wetland in m^2 and Q is the annual loading rate in m^3/yr. These two terms are often combined to give the annual hydraulic loading rate (q):

$$q = \frac{Q}{A} \qquad (5\text{-}8)$$

where, q = annual hydraulic loading rate (m/yr).

$\frac{Q}{A}$ is used in this discussion so that the wetland area can be calculated. As mentioned in the previous section, this is the fundamental difference between the two models. This areal model is based on the premise that treatment in a constructed wetland is based solely on the surface area of the soil-water interface and that increasing depth provides no additional treatment.

k_t is the first-order rate constant and is in units of m/yr. This rate constant may be temperature dependent and the Arrhenius equation is used to model the relationship (Kadlec and Knight, 1996). The Arrhenius equation was described in the previous section (Eq. 5-3).

The three unknowns in this equation are k_{20}, θ, and C^*. As with the volumetric model, k_{20}, θ, and C^* can be solved for simultaneously using nonlinear regression, while minimizing the sum of squared errors between predicted and actual outlet concentrations.

Rearranging Eq. 5-6, the area needed to treat a specific wastewater can be calculated:

$$A = \left(-\frac{Q}{k_t}\right) \ln\left(\frac{C_o - C^*}{C_{in} - C^*}\right). \qquad (5\text{-}9)$$

5.5 Methods

Water quality data was collected at the Oregon State University Dairy Wetland System (OSUDWTS) over a three-year period. These data were used for evaluating treatment performance and to calculate first-order rate constants. Rate constants and fitted coefficients for the first-order models were found for each wetland cell using nonlinear regression while minimizing the sum of squared errors between predicted and actual outlet concentrations. The porosity value was held constant at 0.6 in all regressions conducted to determine volumetric rate constants (see Chapter 4). Average flow in the volumetric model was determined using a detailed water budget for the site (see Chapter 3). The average flow (q) in the areal model was based only on the inlet flows.

5.5.1 Study Site

The OSUDWTS is located in Corvallis, Oregon and was designed to treat diluted dairy flushwater. The site consists of six parallel wetland cells 28.1 m x 5.9 m x 0.30 m (Cells 4-9) and a 29.6 m x 10.7 m x 1.0 m storage pond (Cell 10) (Fig. 5.1). Two of the wetland cells (4 and 9) have a 1 m deep water section in the middle of them that adds approximately 13 m^3 of volume. This deep section is approximately 9 meters long and slopes from 0.3 meters at each end to 1 m in the middle. The wetland system was constructed and planted in 1992, began receiving wastewater in October of 1993, and continues to receive wastewater. Treated water is pumped twice daily from pond 10 to a mixing tank where concentrated dairy wastewater is added. The time of day and duration of the pumping of "recycled water" are controlled by a mechanical timer. The concentrated wastewater is loaded from the dairy's pressurized liquid waste handling system using an electric ball valve and electronic timer. The entire volume of the "mixed" wastewater is then loaded to the cells over a period of approximately four hours. The outflows from the wetland cells drain back into pond 10. The cells are vegetated by a mix of cattails (*Thypha latifolia* L.), bulrush (*Scirpus acutus* Muhl.), and floating grass (Western mannagrass (*Glyceria occidentallis* (Piper) J.C. Nels.) and water foxtail (*Alopercus geniculatus* L.).

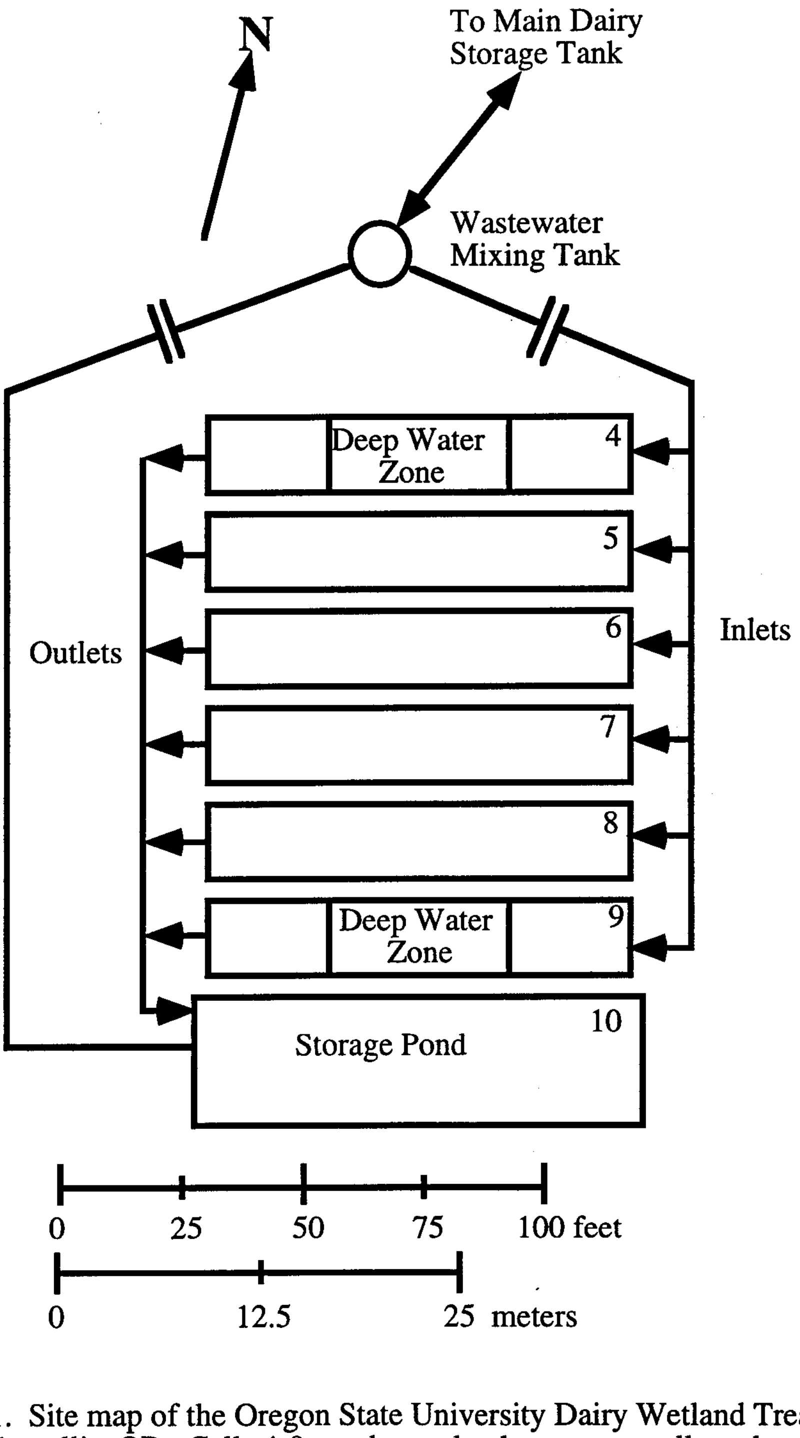

Figure 5.1. Site map of the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Cells 4-9 are the wetland treatment cells and pond 10 is the storage pond. Lines and arrows indicate pipes and the flow path of wastewater.

5.5.2 Wastewater Loading and Sampling

Inlet flows were controlled by 1" ball valves and were checked on each sampling date. If the flows varied from the design flow they were adjusted until design flow was achieved. This procedure was adequate when the flow rates to all wetland cells were equal, however, when flow rates were varied to each of the wetland cells, plugging occurred. A flow splitter, which consisted of a box with six V-notch weirs, was built and installed at the site. After installation of the flow splitter, measured flows were always very close to the design flow. Table 5.2 and 5.3 summarize the wetland dimensions and inlet flow rates during the study period. As mentioned above, cells 4 and 9 have a deep water section and the average depth reported in Table 5.2 is based on the total volume of the wetland divided by the surface area. Daily inlet and outlet flow rates were needed for use in the volumetric based first-order plug flow model and are reported in Table 5.4.

During the first 16 months of operation, the wetlands were all equally loaded with very concentrated dairy wastewater. The theoretical detention time was 8.2 days for wetland cells 5, 6, 7, 8, and 10.2 days for wetland cells 4 and 9 (Table 5.3). Samples of the influent and effluent were collected from all wetland cells once to twice a month. Analyses of fourteen water quality parameters were conducted, which included COD, BOD_5, TS, TSS, TP, PO_4-P, TKN, NH_3, NO_3^-, fecal coliforms, pH, conductivity, dissolved oxygen (DO), and temperature. All water quality analyses were conducted according to Standard Methods (APHA, 1992).

During the spring of 1995 wetlands cells were not loaded with wastewater and were monitored until concentrations reached a steady state. From June 1995 through December 1996, the wetlands were loaded with dilute dairy wastewaters. Wetland cells 4, 5, 6, 7, 8, and 9 were loaded to achieve 10.0, 2.3, 2.3, 8.0, 8.0, and 10.0 day theoretical detention times, respectively (Table 5.3). Water samples were analyzed for BOD_5, TS, TKN, NH_3, pH, DO, and temperature.

Table 5.2. Wetland cell dimensions for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Wetland Cell	Volume (m^3)	Average Depth (m)	Area (m^2)
4	63.2	0.38	166
5	50.5	0.30	166
6	50.5	0.30	166
7	50.5	0.30	166
8	50.5	0.30	166
9	63.2	0.38	166

Table 5.3. Inlet flows and theoretical detention times for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	October 1993 - February 1995			June 1995 - December 1996		
Wetland Cell	Inlet Flow (m^3/d)	Hydraulic Loading Rate (m/yr)	Theoretical Detention Time (d)	Inlet Flow (m^3/d)	Hydraulic Loading Rate (m/yr)	Theoretical Detention Time (d)
4	6.2	13.6	10.2	6.3	13.9	10.0
5	6.2	13.6	8.2	22.1	48.7	2.3
6	6.2	13.6	8.2	22.1	48.7	2.3
7	6.2	13.6	8.2	6.3	13.9	8.0
8	6.2	13.6	8.2	6.3	13.9	8.0
9	6.2	13.6	10.2	6.3	13.9	10.0

Table 5.4. Outlet flows (m^3/d) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. Flows were calculated using calibrated hydrology model (see Chapter 3)

Date	Cell 4	Cell 5	Cell 6	Cell 7	Cell 8	Cell 9
10/22/93	5.1	5.1	4.7	5.3	5.0	5.4
10/27/93	5.3	5.3	5.0	5.5	5.2	5.5
11/4/93	5.4	5.4	5.1	5.5	5.3	5.6
11/17/93	6.9	6.9	6.8	6.9	6.8	6.9
12/2/93	6.1	6.1	6.0	6.1	6.0	6.1
12/16/93	6.0	6.0	6.0	6.1	6.0	6.1
1/10/94	6.2	6.2	6.1	6.2	6.2	6.2
1/17/94	5.9	5.9	5.8	5.9	5.8	5.9
1/31/94	5.6	5.6	5.5	5.7	5.6	5.7
2/7/94	5.7	5.6	5.5	5.7	5.6	5.8
2/14/94	7.0	7.0	6.8	7.0	6.9	7.1
2/28/94	5.9	5.9	5.8	6.0	5.9	6.0
3/14/94	5.5	5.5	5.3	5.7	5.5	5.7
4/8/94	6.5	6.5	6.4	6.6	6.5	6.6
4/20/94	5.3	5.3	4.9	5.5	5.1	5.5
5/4/94	5.6	5.6	5.1	5.8	5.4	5.9
5/18/94	5.2	5.2	4.8	5.4	5.1	5.4
6/9/94	4.2	4.2	3.4	4.6	3.9	4.8
6/15/94	5.1	5.0	4.5	5.3	4.8	5.4
7/21/94	3.4	3.4	2.3	4.0	3.0	4.2
9/12/94	4.8	4.8	4.2	5.0	4.5	5.1
9/29/94	5.6	5.5	5.1	5.8	5.4	5.9
10/13/94	6.1	6.1	5.9	6.2	6.0	6.2
10/27/94	11.6	11.6	11.5	11.6	11.6	11.6
12/1/94	9.2	9.2	9.2	9.2	9.2	9.2
12/14/94	8.2	8.2	8.1	8.2	8.2	8.2
1/11/95	7.3	7.3	7.3	7.3	7.3	7.4
2/16/95	8.4	8.4	8.4	8.4	8.4	8.4
7/27/95	4.3	19.9	19.0	4.7	4.0	4.9
8/16/95	5.7	22.4	22.0	5.9	5.5	6.0
10/10/95	8.2	29.5	29.4	8.3	8.1	8.3
11/21/95	6.3	22.3	22.2	6.3	6.3	6.4
12/11/95	9.6	33.6	33.6	9.6	9.6	9.6
2/1/96	5.8	21.4	21.2	5.9	5.7	5.9
10/4/96	6.2	22.9	22.7	6.3	6.1	6.3
10/11/96	6.0	21.6	21.5	6.0	5.9	6.1
10/18/96	7.3	26.1	26.0	7.4	7.3	7.4
10/25/96	7.1	25.3	25.2	7.1	7.0	7.1
11/1/96	6.1	21.7	21.6	6.1	6.0	6.1
Average =	6.3	11.3	11.0	6.4	6.2	6.5
Std. Dev. =	1.56	8.62	8.68	1.47	1.63	1.44
n =	39	39	39	39	39	39

5.6 Results

Inlet and outlet samples were collected on 39 separate occasions between October of 1993 and November of 1996. A total of 1,526 water quality measurements were conducted. Volumetric and areal constants were determined for COD, BOD, TS, TSS, TP, TKN, NH_3 and fecal coliforms.

5.6.1 Temperature

The temperature of the influent and effluent from each of the wetland cells was measured on most sampling days (Table 5.5). Outlet temperatures were compared to mean daily air temperatures recorded by the Agri-met weather station located at the Hyslop Farms Experimental Station, Corvallis, OR. The correlation between mean daily air temperatures and wetland water temperature was not one to one as suggested by Kadlec and Knight (1996) (Fig. 5.2). The relationship was linear and a line was fit to the data ($R^2 = 0.84$):

$$T_{wet} = 3.19 + 0.63 \cdot T_{air} \qquad (5\text{-}10)$$

where, T_{wet} = wetland temperature (°C) and

T_{air} = mean daily air temperature (°C).

This equation and the mean daily temperature were used to calculate wetland water temperature for all sampling days where water temperature was missing.

5.6.2 Chemical Oxygen Demand

COD was measured on 25 sampling dates during the high waste loading period. The average mass loading rate was 815 kg/ha-d and inlet concentration was 2,181 mg/l.

Table 5.5. Temperature (°C) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Mean Daily Air Temperature	Sample of Inlet	Cell 4	Cell 5	Cell 6	Cell 7	Cell 8	Cell 9
10/22/93	11.2	14.3	---	---	---	---	---	---
10/27/93	10.1	9.7	7.9	7.6	8.2	7.2	7.5	7.4
11/4/93	8.2	9.2	7.3	7.6	7.7	7.7	7.5	8.1
11/17/93	6.0	8.0	6.7	6.7	6.7	6.7	6.7	6.7
12/2/93	7.8	7.6	5.9	6.7	6.7	6.4	6.4	6.2
12/16/93	4.1	8.7	8.7	8.2	8.3	8.4	8.1	8.3
3/14/94	11.1	8.8	7.7	7.6	7.9	8.2	6.6	6.9
4/8/94	10.1	10.2	9.4	---	---	---	---	---
4/20/94	13.3	12.2	---	---	---	12.5	12.2	---
5/4/94	15.4	12.8	---	---	11.8	12.2	---	---
5/18/94	13.5	13.6	---	---	---	---	---	---
7/21/94	27.2	20.0	18.6	18.5	17.9	19.6	18.7	---
9/12/94	15.1	14.1	13.3	13.6	14.7	13.7	11.9	12.1
10/13/94	8.4	9.6	---	---	---	---	---	---
10/27/94	12.8	---	9.1	11.6	11.7	11.4	10.0	6.5
12/1/94	8.0	---	8.0	---	---	8.2	8.2	8.4
12/14/94	4.2	5.4	---	---	---	---	---	---
1/11/95	7.4	---	7.5	7.2	7.2	---	---	7.4
2/16/95	6.4	5.7	4.8	---	---	---	---	---
7/27/95	20.8	19.0	18.3	18.9	18.1	20.3	17.2	17.4
8/16/95	14.9	16.3	15.8	16.0	16.1	15.7	15.1	15.7
10/10/95	14.6	12.8	12.1	12.4	12.4	11.5	12.0	11.9
11/21/95	8.1	7.5	7.7	7.8	7.9	6.8	7.2	7.6
12/11/95	10.6	9.0	7.8	7.7	7.8	7.4	7.1	7.1
2/1/96	-3.1	2.7	1.3	1.2	1.2	3.6	1.3	2.8
10/4/96	16.7	---	13.9	14.0	14.2	13.2	12.8	13.6
10/11/96	14.3	---	13.2	13.8	14.0	13.1	12.9	13.4
10/18/96	8.7	---	10.1	9.5	9.4	8.8	8.5	9.3
10/25/96	8.6	---	9.1	9.3	9.7	8.7	8.8	8.9
11/1/96	5.4	---	6.1	6.1	7.1	7.0	5.1	6.1
Average:	10.7	10.8	9.6	10.1	10.3	10.4	9.6	9.1
Std. Dev.:	5.6	4.3	4.2	4.4	4.2	4.2	4.1	3.6
n:	30	22	24	21	22	23	22	21

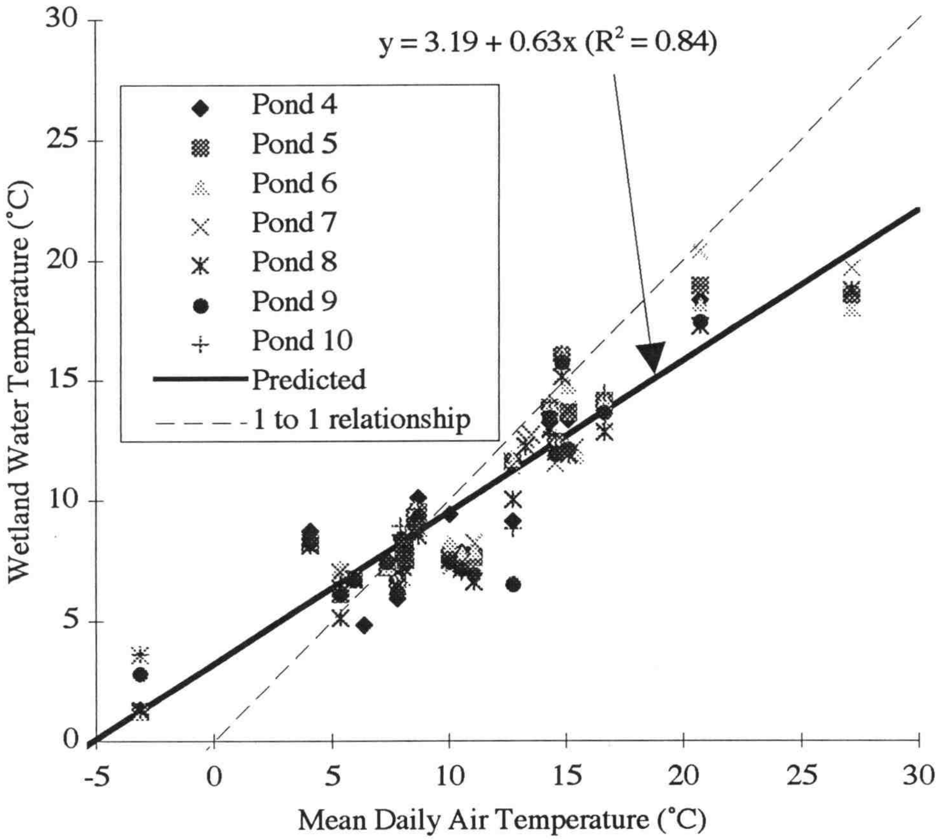

Figure 5.2. Mean daily air temperature versus wetland water temperatures for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

All of the wetland cells had an average reduction between 46% and 48% with the exception of cell 8, which had an average reduction of 37% (Table 5.6). Volumetric and areal first-order rate constants were fit to the data (Fig. 5.3 and 5.4) and are summarized in Table 5.7. The COD loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 10 mg/l for all wetland cells.

5.6.3 Biochemical Oxygen Demand

BOD_5 was measured on 35 sampling dates during the study period. The average mass loading rate was 188 kg/ha-d and inlet concentrations ranged from 39 to 2,211 mg/l with an average of 502 mg/l (Table 5.8). All of the wetland cells had an average reduction between 47% and 56% with a mean of 52%. Volumetric and areal first-order rate constants were fit to the data (Fig. 5.5 and 5.6) and are summarized in Table 5.9. The BOD loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 8.0 mg/l for all wetland cells as suggested by Knight et al. (1996 draft).

5.6.4 Total Solids

TS were measured on 38 sampling dates and the average mass loading was 677 kg/ha-d. The inlet concentration ranged from 276 to 5,744 mg/l with an average of 1,812 mg/l (Table 5.10). Reductions varied from 15 to 31% with an average removal of 27% for all wetland cells. Volumetric and areal first-order rate constants were fit to the data (Fig. 5.7 and 5.8) and are summarized in Table 5.11. The TS loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 20 mg/l for all wetland cells.

Table 5.6. Chemical oxygen demand (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/27/93	549	415	24%	636	-16%	385	30%	412	25%	434	21%	310	44%
11/4/93	510	420	18%	443	13%	398	22%	339	33%	464	9%	379	26%
11/17/93	502	410	18%	347	31%	350	30%	430	14%	391	22%	374	25%
12/2/93	727	83	89%	---	---	---	---	36	95%	151	79%	---	---
12/16/93	816	274	66%	476	42%	617	24%	535	34%	946	-16%	401	51%
1/10/94	551	270	51%	236	57%	170	69%	224	59%	535	3%	289	48%
1/17/94	3,052	604	80%	661	78%	965	68%	885	71%	713	77%	1,128	63%
1/31/94	1,668	653	61%	729	56%	809	52%	835	50%	656	61%	702	58%
2/14/94	1,247	782	37%	739	41%	607	51%	850	32%	1,266	-2%	647	48%
2/28/94	2,688	336	88%	547	80%	467	83%	383	86%	453	83%	375	86%
3/14/94	1,122	568	49%	593	47%	646	42%	950	15%	724	35%	686	39%
4/20/94	3,425	1,025	70%	842	75%	1,220	64%	1,441	58%	1,169	66%	1,068	69%
5/4/94	3,459	---	---	752	78%	975	72%	995	71%	---	---	---	---
5/18/94	4,161	---	---	1,112	73%	1,228	70%	1,060	75%	---	---	---	---
6/9/94	3,804	---	---	1,016	73%	1,104	71%	1,198	69%	---	---	---	---
6/15/94	1,965	---	---	1,074	45%	1,160	41%	1,128	43%	---	---	---	---
7/21/94	3,366	1,533	54%	1,692	50%	1,553	54%	1,481	56%	1,449	57%	---	---
9/12/94	2,005	1,185	41%	1,107	45%	1,032	49%	888	56%	921	54%	873	56%
9/29/94	1,455	1,050	28%	1,221	16%	1,113	24%	1,233	15%	981	33%	897	38%
10/13/94	3,033	1,422	53%	1,632	46%	1,482	51%	1,959	35%	1,659	45%	1,080	64%
10/27/94	3,708	2,418	35%	2,607	30%	2,316	38%	2,268	39%	2,673	28%	2,130	43%
12/1/94	4,370	2,634	40%	2,478	43%	2,550	42%	2,616	40%	2,844	35%	2,601	40%
12/14/94	2,146	1,497	30%	1,383	36%	1,209	44%	1,272	41%	1,500	30%	1,464	32%
1/11/95	1,860	1,416	24%	693	63%	1,200	35%	1,461	21%	1,200	35%	1,575	15%
2/16/95	2,345	1,680	28%	1,638	30%	1,629	31%	1,806	23%	1,674	29%	1,647	30%
Average:	2,181	984	47%	1,027	47%	1,049	48%	1,067	46%	1,086	37%	980	46%
Std. Dev.:	1,257	705	22%	618	24%	584	18%	640	23%	708	27%	653	17%
n:	25	21	21	24	24	24	24	25	25	21	21	19	19

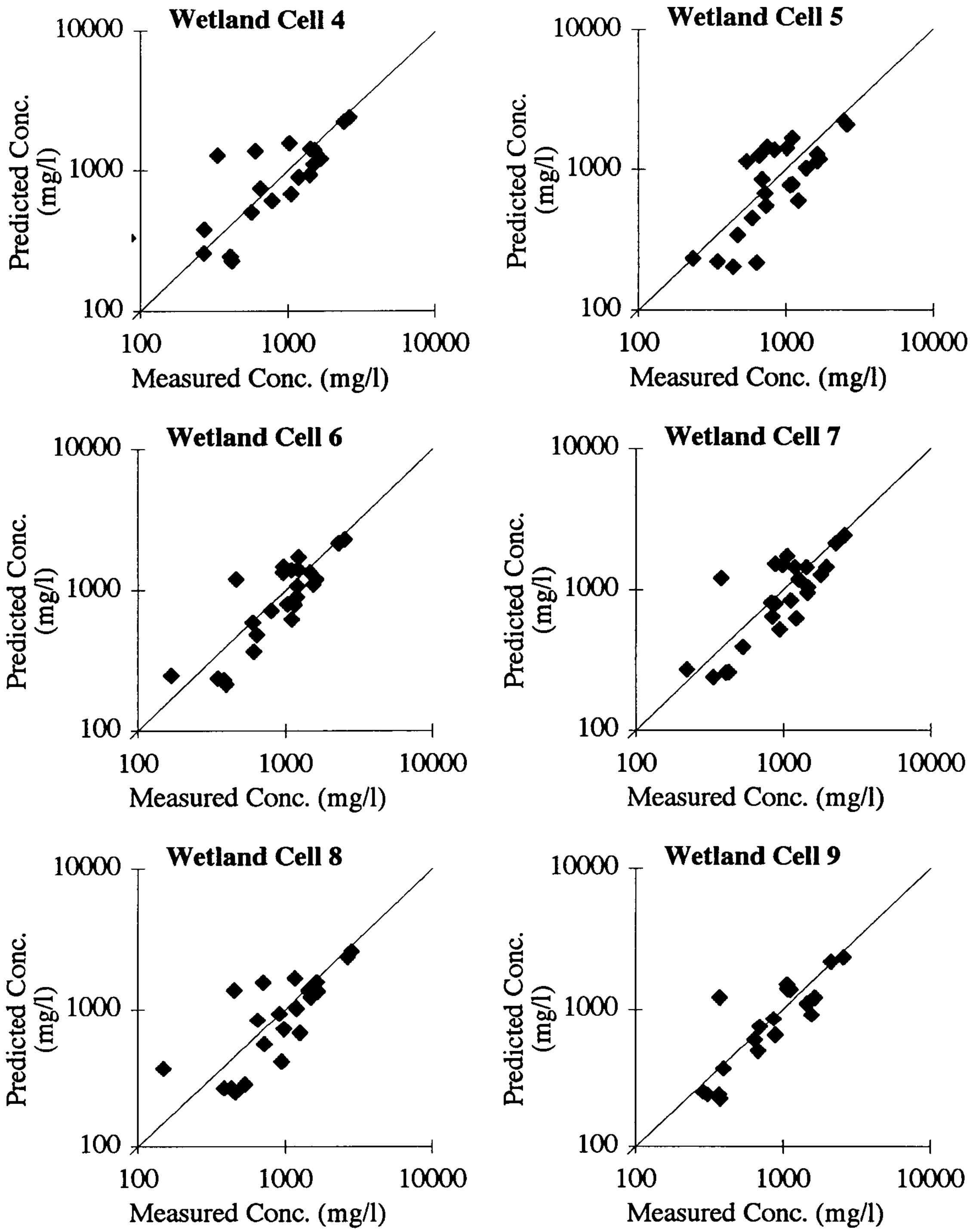

Figure 5.3. Scatterplot of measured versus predicted chemical oxygen demand concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

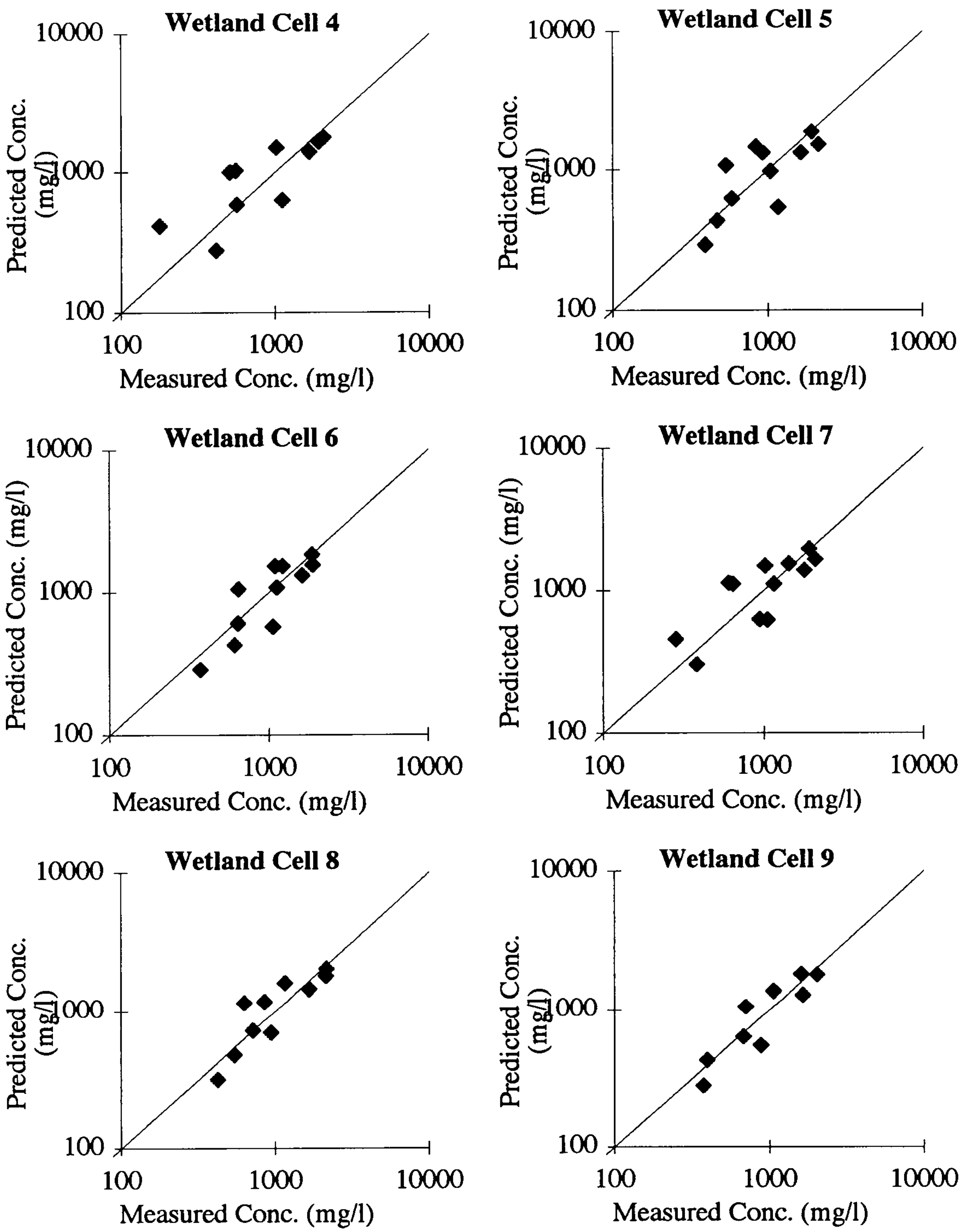

Figure 5.4. Scatterplot of measured versus predicted chemical oxygen demand concentrations using the k-C* model with C* held at 10 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.7. Chemical oxygen demand rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n[a]	k_{20}	θ	R^2	C^{*b}	k_{20}	θ	R^2
Pond 4	0.60	0.11	0.99	0.71	10	25	1.09	0.70
Pond 5	0.60	0.13	1.00	0.59	10	37	1.13	0.54
Pond 6	0.60	0.13	1.00	0.72	10	28	1.10	0.69
Pond 7	0.60	0.16	1.02	0.64	10	30	1.12	0.65
Pond 8	0.60	0.11	1.00	0.71	10	32	1.13	0.78
Pond 9	0.60	0.13	1.00[c]	0.75	10	41	1.13	0.81
Average =	0.60	0.13	1.00	0.69	10	32	1.12	0.69
Std. Dev. =	0.00	0.02	0.01	0.06	0.0	5.9	0.02	0.10

[a] n held constant at 0.6

[b] C^* held constant at 10 mg/l

[c] θ held constant at 1.00

Table 5.8. Biochemical oxygen demand (5 day) (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	162	---	---	145	11%	74	54%	106	35%	110	32%	---	---
10/27/93	168	131	22%	176	-4%	116	31%	119	29%	119	29%	80	53%
11/4/93	155	120	22%	113	27%	87	44%	122	21%	122	21%	96	38%
11/17/93	139	60	57%	60	57%	---	---	83	40%	70	49%	63	55%
12/2/93	265	73	72%	---	---	---	---	44	83%	75	72%	---	---
12/16/93	209	85	59%	160	23%	215	-3%	179	14%	334	-60%	137	34%
1/10/94	149	52	65%	40	73%	27	82%	35	77%	157	-6%	61	59%
1/17/94	928	133	86%	156	83%	266	71%	245	74%	166	82%	326	65%
1/31/94	574	193	66%	181	69%	280	51%	245	57%	141	75%	190	67%
2/14/94	347	179	48%	154	56%	129	63%	224	35%	350	-1%	125	64%
2/28/94	788	85	89%	124	84%	84	89%	111	86%	95	88%	65	92%
3/14/94	447	171	62%	161	64%	189	58%	344	23%	218	51%	227	49%
4/8/94	1,475	170	88%	220	85%	---	---	428	71%	480	67%	273	82%
4/20/94	1,268	321	75%	187	85%	438	65%	383	70%	363	71%	339	73%
5/4/94	1,679	---	---	134	92%	221	87%	228	86%	---	---	---	---
6/9/94	2,211	---	---	341	85%	430	81%	640	71%	---	---	---	---
6/15/94	1,080	---	---	369	66%	415	62%	428	60%	---	---	---	---
7/21/94	813	288	65%	328	60%	266	67%	232	71%	254	69%	---	---
9/29/94	494	188	62%	266	46%	233	53%	286	42%	204	59%	112	77%
10/13/94	644	200	69%	256	60%	207	68%	216	66%	316	51%	105	84%
10/27/94	803	592	26%	575	28%	460	43%	535	33%	721	10%	529	34%
12/1/94	439	275	37%	74	83%	168	62%	288	34%	300	32%	271	38%
12/14/94	850	392	54%	337	60%	338	60%	390	54%	390	54%	427	50%
2/16/95	837	566	32%	536	36%	525	37%	683	18%	677	19%	502	40%
7/27/95	84	113	-34%	27	68%	70	17%	96	-15%	60	28%	64	24%
8/16/95	41	52	-25%	49	-19%	39	7%	24	42%	17	59%	20	53%
10/10/95	56	25	56%	82	-46%	42	25%	29	48%	19	65%	29	48%
11/21/95	87	58	34%	68	22%	57	35%	54	38%	39	55%	63	28%
12/11/95	61	26	57%	30	50%	25	59%	32	47%	27	55%	39	36%
2/1/96	57	34	40%	25	56%	26	55%	11	81%	28	52%	29	50%
10/4/96	48	14	72%	9	81%	11	76%	15	68%	13	73%	11	78%
10/11/96	43	15	66%	12	73%	13	69%	20	54%	15	64%	19	57%
10/18/96	39	7	83%	19	52%	19	51%	14	64%	13	67%	13	68%
10/25/96	54	27	50%	37	32%	35	35%	19	65%	21	62%	27	49%
11/1/96	92	32	65%	31	66%	33	64%	27	71%	26	72%	26	72%
Average:	502	151	52%	161	52%	173	54%	198	52%	186	47%	147	56%
Std. Dev.:	537	151	29%	144	32%	154	22%	185	24%	189	31%	151	18%
n:	35	31	31	34	34	32	32	35	35	32	32	29	29

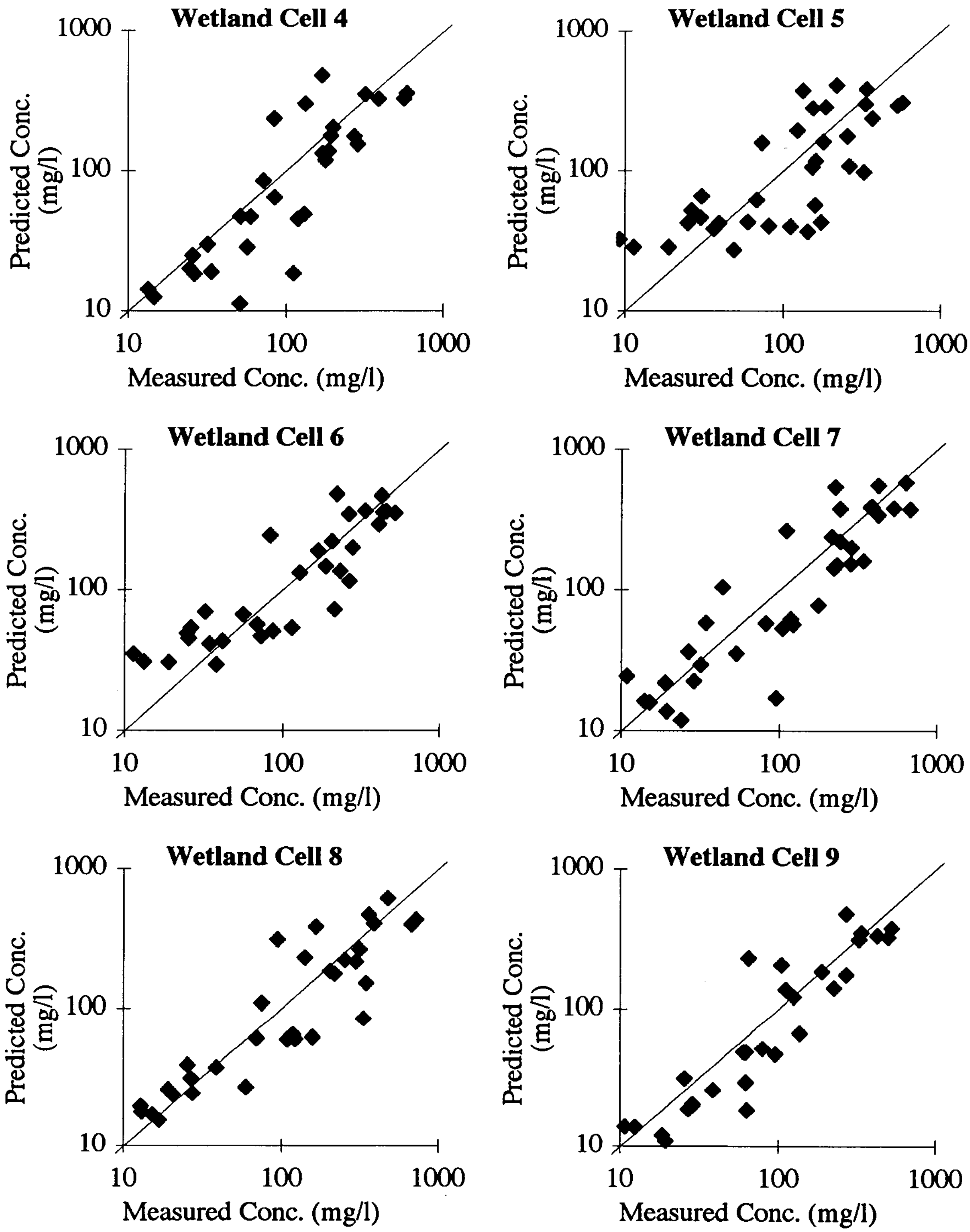

Figure 5.5. Scatterplot of measured versus predicted BOD_5 concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

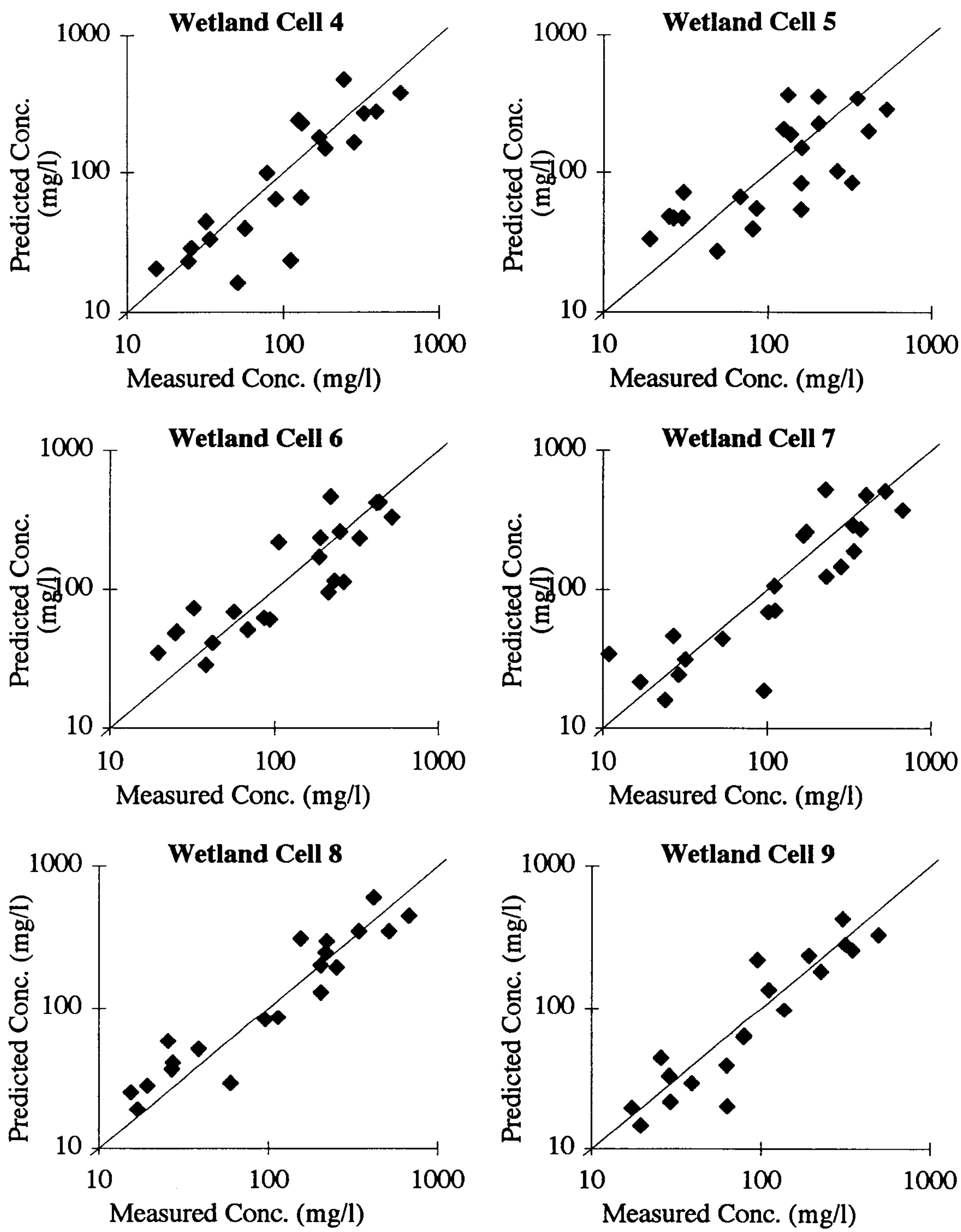

Figure 5.6. Scatterplot of measured versus predicted BOD_5 concentrations using k-C^* model with C^* held at 8 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.9. Biochemical oxygen demand (5 day) rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n^a	k_{20}	θ	R^2	C^{*b}	k_{20}	θ	R^2
Pond 4	0.60	0.22	1.01	0.57	8.0	24	1.05	0.63
Pond 5	0.60	0.28	1.02	0.44	8.0	36	1.07	0.34
Pond 6	0.60	0.23	1.03	0.69	8.0	32	1.07	0.65
Pond 7	0.60	0.23	1.03	0.70	8.0	27	1.07^c	0.64
Pond 8	0.60	0.16	1.01	0.63	8.0	22	1.07	0.76
Pond 9	0.60	0.24	1.02	0.74	8.0	30	1.07^c	0.78
Average =	0.60	0.23	1.02	0.63	8.0	29	1.07	0.63
Std. Dev. =	0.00	0.04	0.01	0.11	0.0	5.2	0.01	0.16

[a] n held constant at 0.6

[b] C^* held constant at 8 mg/l

[c] θ held constant at 1.07

Table 5.10. Total solids (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	612	310	49%	631	-3%	342	44%	486	21%	533	13%	341	44%
10/27/93	612	492	20%	797	-30%	512	16%	475	22%	523	14%	418	32%
11/4/93	682	582	15%	616	10%	576	16%	502	26%	634	7%	584	14%
11/17/93	724	534	26%	476	34%	500	31%	494	32%	534	26%	884	-22%
12/2/93	1,124	420	63%	328	71%	219	81%	284	75%	470	58%	248	78%
12/16/93	888	468	47%	636	28%	730	18%	620	30%	1,048	-18%	492	45%
1/10/94	678	416	39%	336	50%	252	63%	328	52%	632	7%	380	44%
1/17/94	2,358	678	71%	716	70%	1,226	48%	902	62%	788	67%	1,156	51%
1/31/94	1,624	878	46%	860	47%	940	42%	888	45%	808	50%	826	49%
2/14/94	1,170	926	21%	870	26%	752	36%	918	22%	1,258	-8%	740	37%
2/28/94	2,320	536	77%	606	74%	402	83%	458	80%	462	80%	342	85%
3/14/94	1,246	732	41%	788	37%	802	36%	1,036	17%	994	20%	852	32%
4/8/94	3,700	860	77%	960	74%	1,420	62%	1,340	64%	1,680	55%	1,040	72%
4/20/94	2,836	1,290	55%	1,188	58%	1,422	50%	1,398	51%	1,372	52%	1,312	54%
5/4/94	3,994	---	---	1,076	73%	1,202	70%	1,218	70%	---	---	---	---
5/18/94	5,168	---	---	1,584	69%	1,690	67%	1,600	69%	---	---	---	---
6/9/94	5,744	---	---	2,060	64%	2,120	63%	2,260	61%	---	---	---	---
6/15/94	3,372	---	---	2,052	39%	2,168	36%	2,090	38%	---	---	---	---
7/21/94	2,836	1,846	35%	2,176	23%	2,032	28%	1,966	31%	1,918	32%	---	---
9/12/94	2,554	2,158	16%	2,178	15%	2,046	20%	1,828	28%	1,864	27%	1,820	29%
9/29/94	2,468	1,998	19%	2,162	12%	2,104	15%	2,226	10%	2,026	18%	1,882	24%
10/13/94	3,820	2,526	34%	2,688	30%	2,550	33%	2,733	28%	2,788	27%	2,118	45%
10/27/94	5,272	3,464	34%	3,730	29%	3,460	34%	3,364	36%	3,744	29%	3,168	40%
12/1/94	1,420	1,700	-20%	870	39%	1,334	6%	1,752	-23%	1,916	-35%	1,640	-15%
12/14/94	2,056	1,752	15%	1,868	9%	1,610	22%	1,668	19%	2,446	-19%	2,124	-3%
1/11/95	1,915	1,698	11%	908	53%	1,412	26%	1,624	15%	1,402	27%	1,748	9%
2/16/95	2,397	1,998	17%	1,820	24%	1,876	22%	2,016	16%	1,978	17%	1,950	19%
7/27/95	519	406	22%	586	-13%	552	-6%	768	-48%	968	-86%	538	-4%
8/16/95	376	320	15%	500	-33%	362	4%	324	14%	606	-61%	408	-9%
10/10/95	597	452	24%	908	-52%	506	15%	456	24%	514	14%	478	20%
11/21/95	586	510	13%	618	-5%	668	-14%	558	5%	564	4%	514	12%
12/11/96	446	345	23%	384	14%	349	22%	335	25%	368	17%	401	10%
2/1/96	346	448	-30%	268	22%	310	10%	192	44%	340	2%	284	18%
10/4/96	487	356	27%	320	34%	323	34%	381	22%	373	23%	345	29%
10/11/96	276	239	13%	248	10%	201	27%	263	5%	291	-5%	276	0%
10/18/96	519	265	49%	447	14%	475	8%	435	16%	441	15%	372	28%
10/25/96	496	360	27%	441	11%	452	9%	329	34%	403	19%	372	25%
11/1/96	601	533	11%	540	10%	493	18%	475	21%	505	16%	461	23%
Average:	1,812	956	29%	1,059	27%	1,063	31%	1,079	30%	1,094	15%	925	28%
Std. Dev.:	1,526	793	23%	799	31%	793	23%	804	26%	828	33%	732	25%
n:	38	34	34	38	38	38	38	38	38	34	34	33	33

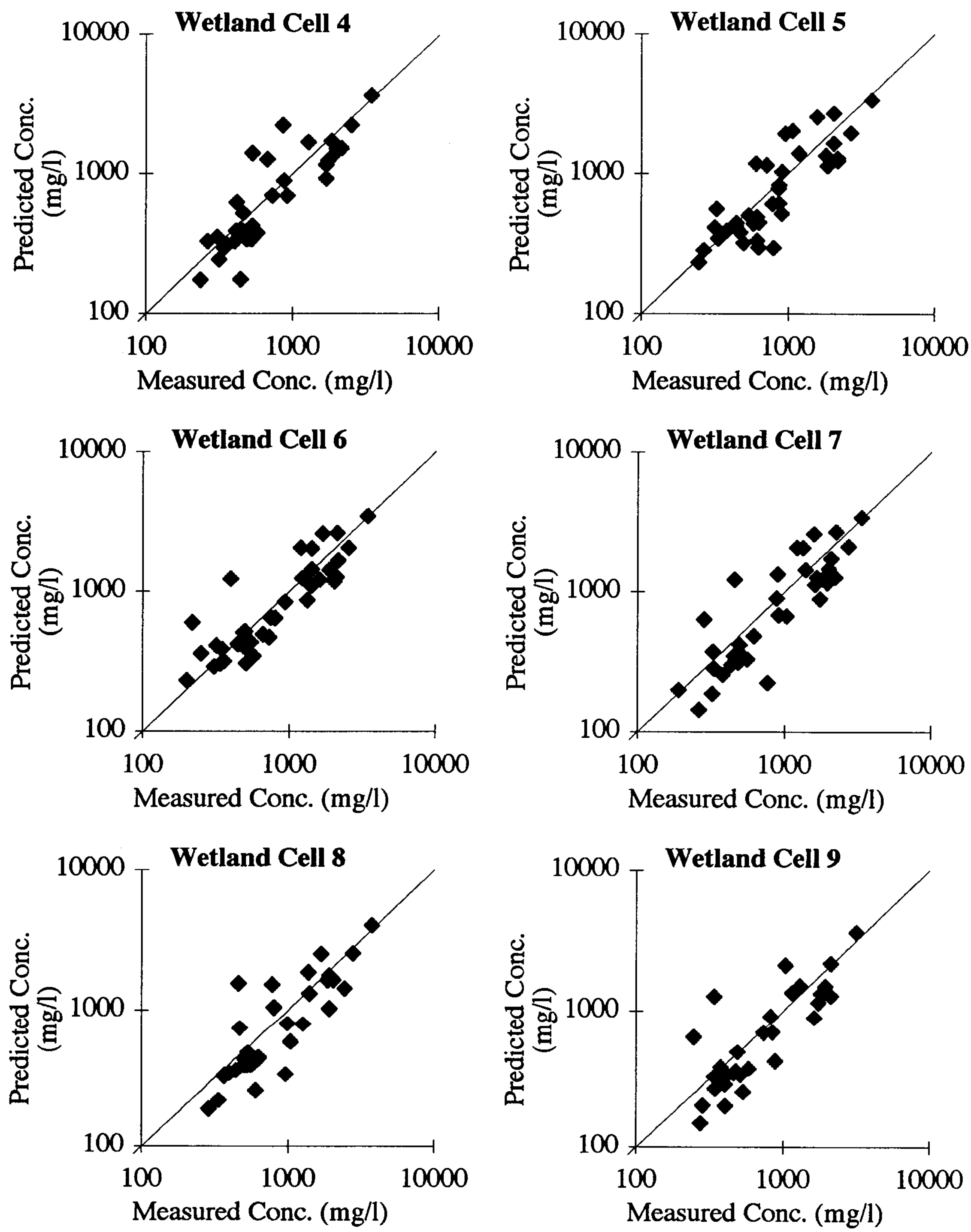

Figure 5.7. Scatterplot of measured versus predicted total solids concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

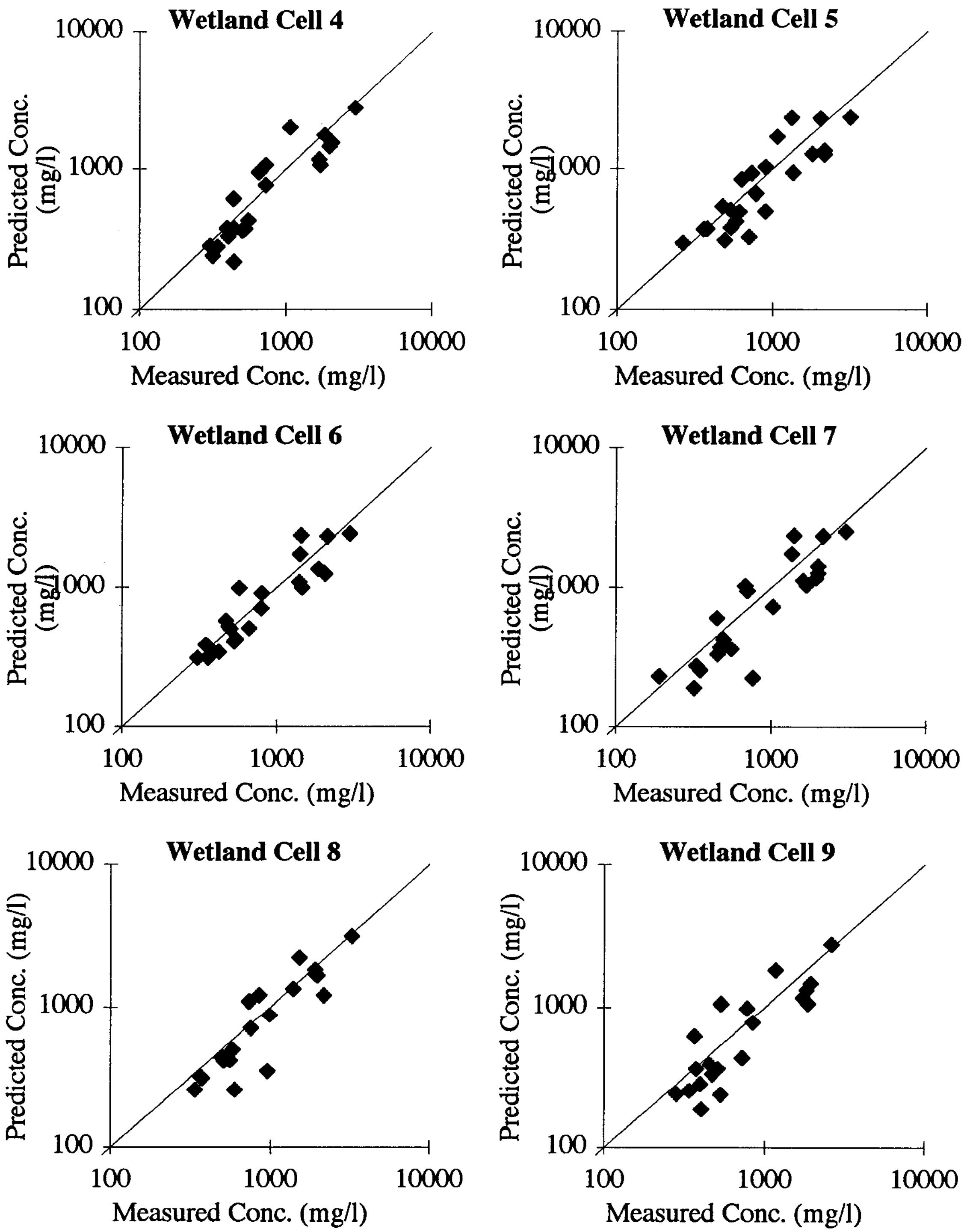

Figure 5.8. Scatterplot of measured versus predicted total solids concentrations using the k-C^* model with C^* held at 20 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.11. Total solids rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n[a]	k_{20}	θ	R^2	C^{*b}	k_{20}	θ	R^2
Pond 4	0.60	0.06	0.97	0.74	20	6.6	1.00	0.79
Pond 5	0.60	0.10	0.99	0.66	20	11	1.02	0.66
Pond 6	0.60	0.10	1.00	0.74	20	12	1.03	0.74
Pond 7	0.60	0.12	1.02	0.71	20	13	1.04	0.72
Pond 8	0.60	0.06	0.99	0.76	20	6.4	1.02	0.83
Pond 9	0.60	0.11	1.01	0.73	20	13	1.05	0.75
Average =	0.60	0.09	1.00	0.72	20	10	1.03	0.75
Std. Dev. =	0.00	0.03	0.02	0.03	0	3.0	0.02	0.06

[a] n held constant at 0.6

[b] C^* held constant at 20 mg/l

5.6.5 Total Suspended Solids

TSS were measured on 26 sampling dates during the high loading period. Inlet concentrations varied from 75 to 1,705 mg/l with an average of 542 mg/l. The average mass loading rate was 203 kg/ha-d. Concentrations were reduced in all wetland cells by an average of 55%, with a range from 43 to 60% (Table 5.12). Table 5.13 is a summary of the volumetric and areal rate constants that were fit to the data (Fig. 5.9 and 5.10). The TSS loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 10 mg/l for all wetland cells.

5.6.6 Total Phosphorus

TP was measured on 33 sampling dates during the high loading period. The average mass loading was 12 kg/ha-d and inlet concentrations ranged from 3 to 115 mg/l with an average of 33 mg/l (Table 5.14). The range of reductions in the wetland cells was 33 to 48% with mean of 42%. Volumetric and areal rate constants are summarized in Table 5.15 and plots of the data are shown in Fig. 5.11 and 5.12. The TP loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 1.0 mg/l for all wetland cells.

5.6.7 Orthophosphate

PO_4-P was measured on seven sampling days during the start-up phase. Average mass loading was 1.4 kg/ha-d and inlet concentrations varied from 1.2 to 11 mg/l (Table 5.16). The average reduction in the wetland cells was 43% with a range from 20 to 59%. Rate constants could not be found given the limited data set.

Table 5.12. Total suspended solids (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	165	66	60%	194	-18%	73	56%	56	66%	151	8%	61	63%
10/27/93	134	74	44%	238	-79%	113	16%	84	37%	110	18%	48	64%
11/4/93	92	67	28%	81	13%	86	7%	44	52%	111	-20%	53	43%
11/17/93	86	35	60%	30	65%	27	69%	76	12%	55	37%	54	38%
12/2/93	75	33	56%	30	60%	43	43%	63	16%	45	40%	38	50%
12/16/93	133	35	74%	89	33%	76	43%	104	22%	100	25%	134	-1%
1/10/94	104	42	60%	83	21%	14	87%	22	79%	35	66%	74	29%
1/17/94	470	101	79%	186	61%	496	-6%	68	86%	64	86%	248	47%
1/31/94	211	83	61%	46	78%	41	81%	48	77%	40	81%	35	83%
2/14/94	157	50	68%	62	61%	47	70%	58	63%	71	55%	35	78%
2/28/94	1,456	38	97%	45	97%	22	98%	40	97%	70	95%	25	98%
3/14/94	304	56	82%	45	85%	43	86%	96	68%	241	21%	70	77%
4/8/94	245	69	72%	87	64%	173	29%	199	19%	363	-48%	177	28%
4/20/94	876	244	72%	224	74%	248	72%	338	61%	426	51%	278	68%
5/4/94	756	---	---	90	88%	95	87%	120	84%	---	---	---	---
5/18/94	1,172	---	---	58	95%	98	92%	69	94%	---	---	---	---
6/9/94	1,705	---	---	107	94%	120	93%	147	91%	---	---	---	---
6/15/94	836	---	---	92	89%	84	90%	136	84%	---	---	---	---
7/21/94	924	260	72%	340	63%	228	75%	276	70%	---	---	---	---
9/12/94	704	65	91%	99	86%	36	95%	47	93%	51	93%	4	99%
9/29/94	374	125	67%	186	50%	157	58%	196	48%	147	61%	46	88%
10/13/94	906	121	87%	203	78%	106	88%	256	72%	172	81%	37	96%
10/27/94	932	150	84%	129	86%	82	91%	97	90%	127	86%	126	86%
12/14/94	352	454	-29%	538	-53%	272	23%	480	-36%	373	-6%	410	-16%
1/11/95	475	415	13%	158	67%	303	36%	263	45%	213	55%	365	23%
2/16/95	457	368	20%	368	20%	354	23%	440	4%	396	13%	448	2%
Average:	542	134	60%	146	53%	132	62%	147	57%	160	43%	132	54%
Std. Dev.:	450	129	29%	120	45%	118	31%	125	34%	127	40%	136	34%
n:	26	22	22	26	26	26	26	26	26	21	21	21	21

Table 5.13. Total suspended solids rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n[a]	k_{20}	θ[b]	R^2	C^*[c]	k_{20}	θ[b]	R^2
Pond 4	0.60	0.28	1.01	0.07	20	25	1.01	0.05
Pond 5	0.60	0.31	1.01	0.00	20	28	1.01	0.00
Pond 6	0.60	0.31	1.01	0.01	20	30	1.01	0.00
Pond 7	0.60	0.30	1.01	0.03	20	27	1.01	0.01
Pond 8	0.60	0.25	1.01	0.03	20	23	1.01	0.02
Pond 9	0.60	0.31	1.01	0.03	20	27	1.01	0.00
Average =	0.60	0.29	1.01	0.03	20	27	1.01	0.01
Std. Dev. =	0.00	0.02	0.00	0.02	0.0	2.6	0.00	0.02

[a] n held constant at 0.6

[b] θ held constant at 1.01

[c] C^* held constant at 20 mg/l

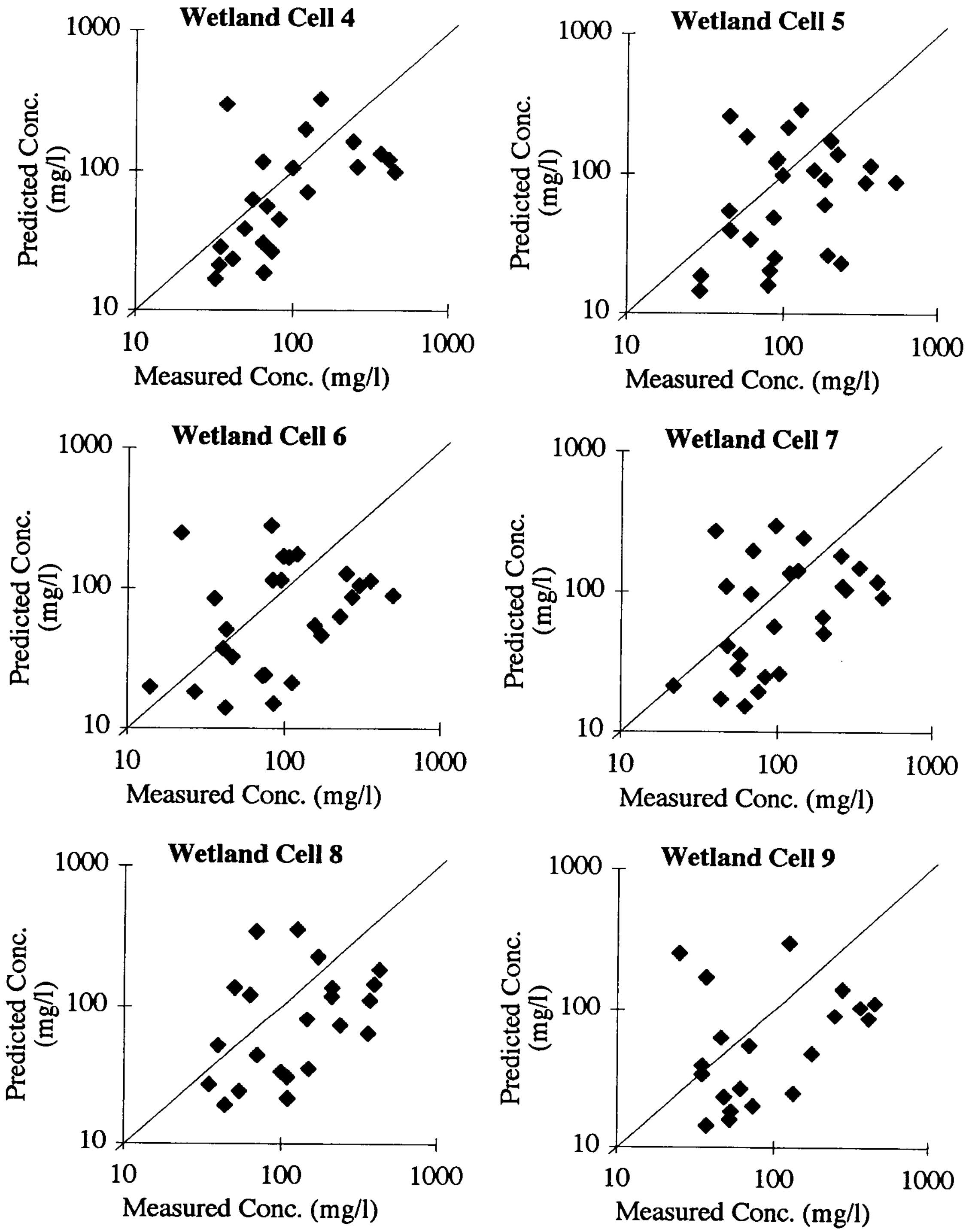

Figure 5.9. Scatterplot of measured versus predicted total suspended solids concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. n and θ held constant at 0.60 and 1.01, respectively.

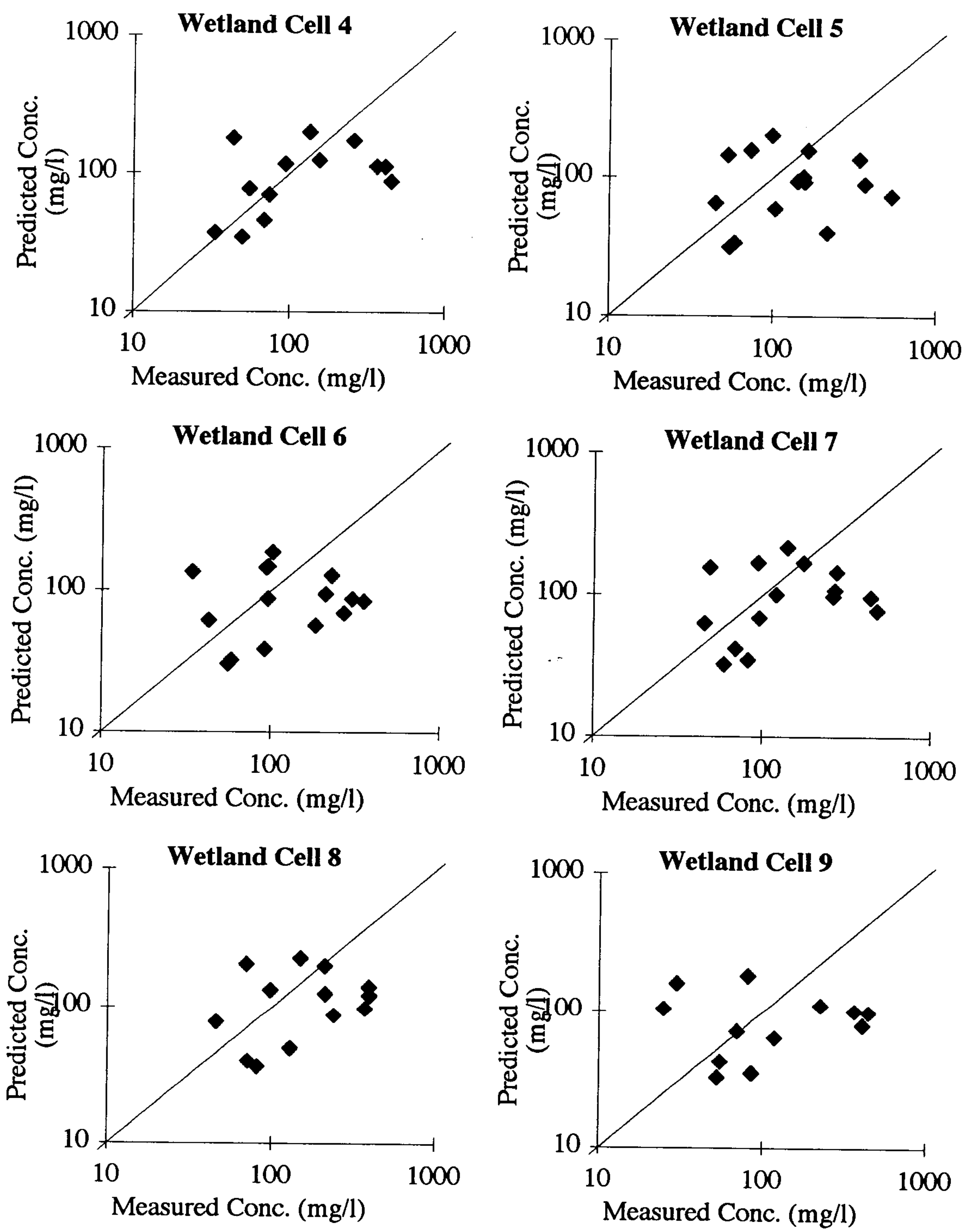

Figure 5.10. Scatterplot of measured versus predicted total suspended solids using the k-C* model with C* held at 20 mg/l and θ at 1.01 for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.14. Total phosphorus (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	5	1	76%	3	33%	1	75%	4	21%	3	44%	2	47%
11/17/93	3	3	12%	3	16%	3	28%	3	22%	3	3%	3	28%
12/2/93	6	3	59%	1	87%	1	91%	2	63%	4	45%	1	88%
1/17/94	23	5	77%	4	83%	3	89%	8	66%	8	67%	5	79%
1/31/94	20	10	51%	10	49%	11	45%	10	52%	8	61%	9	56%
2/14/94	14	10	29%	8	41%	7	50%	10	30%	15	-11%	7	49%
2/28/94	30	5	84%	6	79%	4	86%	5	82%	5	83%	4	86%
3/14/94	11	5	53%	5	58%	6	44%	9	17%	6	44%	6	44%
4/8/94	41	6	87%	7	84%	18	58%	13	69%	16	61%	8	81%
4/20/94	9	7	29%	5	44%	6	35%	6	36%	7	28%	6	40%
5/18/94	56	---	---	12	78%	14	75%	12	79%	---	---	---	---
6/9/94	61	---	---	22	64%	23	62%	26	57%	---	---	---	---
6/15/94	34	---	---	24	29%	25	26%	25	26%	---	---	---	---
7/21/94	46	22	52%	28	39%	27	41%	27	41%	23	---	---	---
9/12/94	31	30	3%	31	0%	29	5%	27	14%	28	11%	21	31%
9/29/94	36	29	19%	33	8%	29	18%	35	3%	29	18%	23	36%
10/13/94	60	38	37%	38	37%	37	38%	40	33%	40	33%	25	58%
10/27/94	115	46	60%	46	60%	47	59%	48	58%	51	55%	44	61%
12/1/94	29	27	8%	21	29%	23	21%	29	0%	30	-4%	26	10%
12/14/94	31	30	5%	30	4%	27	13%	28	11%	30	3%	28	10%
1/11/95	28	22	21%	14	50%	15	46%	18	36%	17	39%	19	32%
2/16/95	30	25	18%	21	31%	19	38%	27	11%	26	15%	28	8%
Average:	33	17	41%	17	46%	17	48%	19	38%	18	33%	15	47%
Std. Dev.:	25	14	28%	13	27%	13	25%	13	25%	14	27%	12	26%
n:	22	19	19	22	22	22	22	22	22	19	18	18	18

Table 5.15. Total phosphorus rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n^a	k_{20}	θ	R^2	C^{*c}	k_{20}	θ	R^2
Pond 4	0.60	0.10	0.98	0.58	1.0	11	1.01	0.76
Pond 5	0.60	0.12	0.99	0.54	1.0	12	1.01	0.66
Pond 6	0.60	0.11	0.98	0.62	1.0	12	1.01	0.73
Pond 7	0.60	0.14	1.02	0.58	1.0	13	1.04	0.64
Pond 8	0.60	0.11	1.00	0.69	1.0	12	1.03	0.83
Pond 9	0.60	0.14	0.99^b	0.64	1.0	13	1.02^d	0.68
Average =	0.60	0.12	0.99	0.61	1.0	12	1.02	0.72
Std. Dev. =	0.00	0.02	0.01	0.05	0.0	0.90	0.01	0.07

[a] n held constant at 0.6

[b] θ held constant at 0.99

[c] C^* held constant at 20 mg/l

[d] θ held constant at 1.02

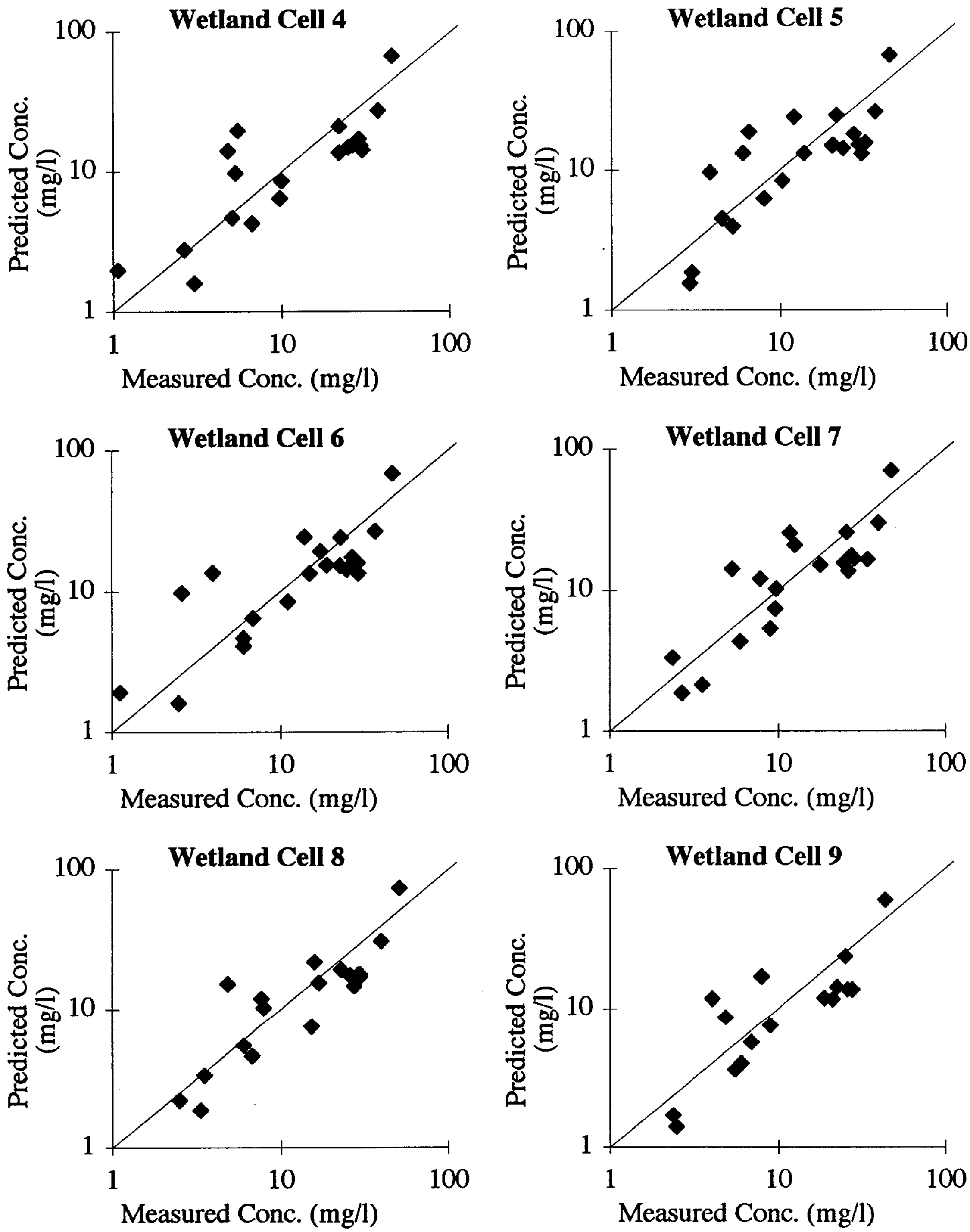

Figure 5.11. Scatterplot of measured versus predicted total phosphorus concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. n held constant at 0.6.

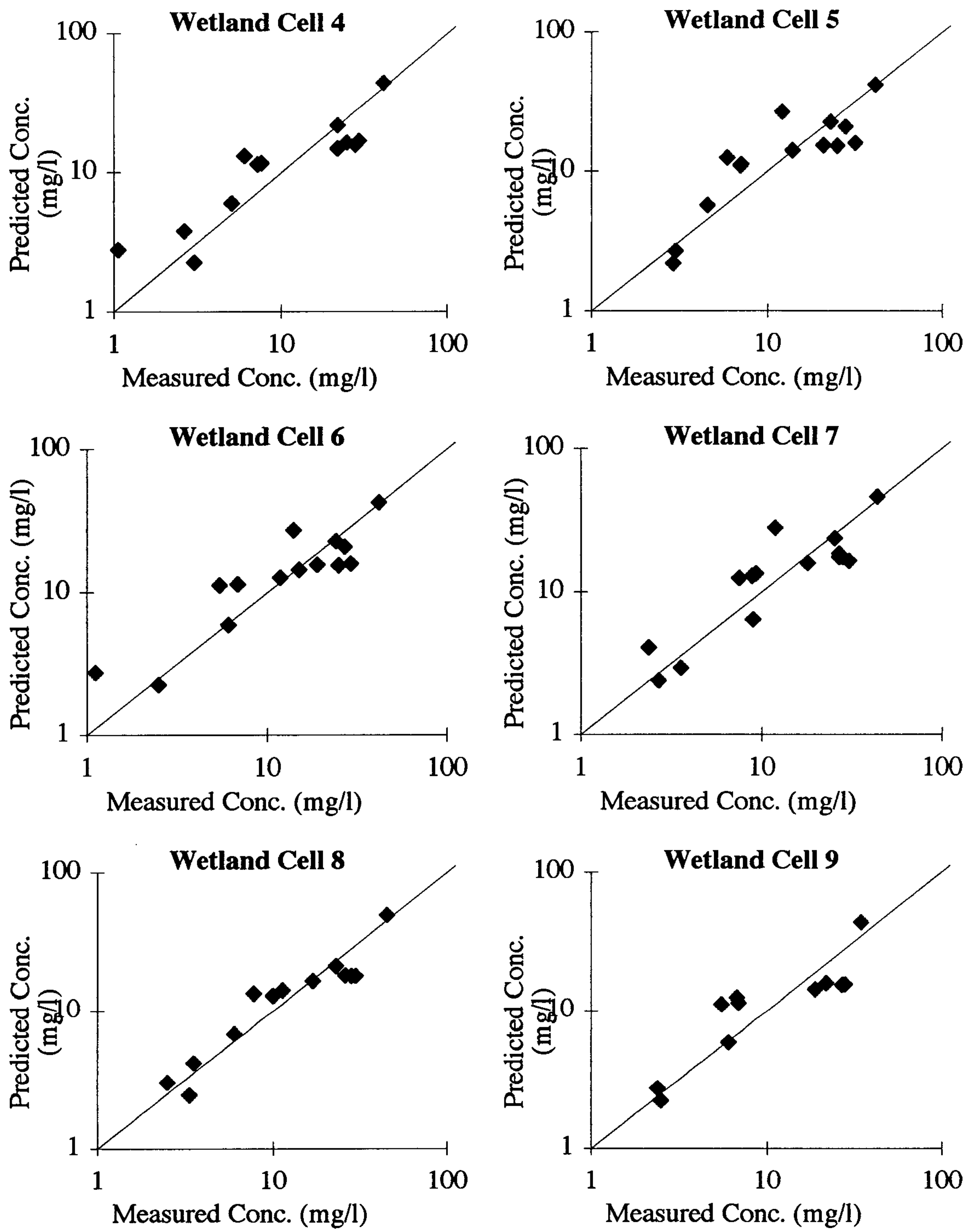

Figure 5.12. Scatterplot of measured versus predicted total phosphorus concentrations using the k-C* model with C* held at 1 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.16. Orthophosphate (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	3.6	0.7	81%	2.3	37%	0.5	87%	2.5	32%	1.4	60%	1.5	59%
10/27/93	3.2	2.4	25%	4.3	-33%	2.1	34%	2.1	36%	2.4	24%	1.8	46%
11/4/93	2.8	2.3	17%	2.0	27%	1.7	38%	1.2	58%	2.9	-4%	2.0	26%
11/17/93	1.2	1.2	-3%	1.0	16%	0.8	30%	0.8	33%	1.1	6%	0.9	24%
12/2/93	2.4	1.1	57%	0.4	85%	0.2	90%	0.9	62%	1.4	43%	0.3	86%
1/10/94	2.7	1.3	52%	0.7	74%	0.8	71%	1.1	58%	4.0	-48%	1.4	48%
1/17/94	11.0	3.2	71%	2.6	77%	4.1	63%	4.7	57%	4.7	57%	5.6	49%
Average:	3.8	1.7	43%	1.9	40%	1.5	59%	1.9	48%	2.6	20%	1.9	48%
Std. Dev.:	3.2	0.9	30%	1.4	42%	1.3	25%	1.4	14%	1.4	39%	1.7	21%
n:	7	7	7	7	7	7	7	7	7	7	7	7	7

5.6.8 Total Kjeldahl Nitrogen

TKN was measured on 30 sampling dates. The average mass loading was 55 kg/ha-d and the average inlet concentration ranged from 16 to 417 mg/l with a mean of 148 mg/l (Table 5.17). The average reduction in TKN was 41% with a range from 33 to 46%. Table 5.18 is a summary of the first-order rate constants fit to the data and plots of the data are shown in Fig. 5.13 and 5.14. The TKN loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 10 mg/l for all wetland cells.

5.6.9 Ammonia

Samples were analyzed for NH_3 on 38 sampling dates. The average NH_3 mass loading was 35 kg/ha-d and the concentrations ranged from 7 to 301 mg/l with a mean of 93 mg/l (Table 5.19). The average reduction in NH_3 was 37% with a range from 31 to 44%. Rate constants were fit to the data and are summarized in Table 5.20 plots of the data are shown in Fig. 5.15 and 5.16. The NH_3 loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 3.0 mg/l for all wetland cells.

5.6.10 Nitrate

NO_3^- measurements were made on five sampling dates during the start-up period. Inlet concentrations varied from 0.00 to 0.66 mg/l with a mean of 0.18 mg/l (Table 5.21). Outlet NO_3^- concentrations were generally lower then inlet concentrations (16% reduction). Rate constants were not fit to the data set because of its limited size.

Table 5.17. Total Kjeldahl nitrogen (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	43	15	65%	38	13%	14	68%	24	45%	29	32%	17	61%
10/27/93	99	31	69%	45	55%	24	76%	25	74%	32	68%	23	77%
11/4/93	38	31	19%	34	11%	29	22%	24	37%	35	7%	29	24%
11/17/93	42	29	29%	26	37%	26	37%	27	35%	31	26%	27	36%
12/2/93	67	23	66%	12	82%	9	86%	17	75%	32	52%	11	83%
12/16/93	76	29	62%	46	40%	60	21%	50	34%	88	-15%	33	56%
1/10/94	53	35	34%	20	63%	20	63%	26	51%	55	-4%	27	49%
1/17/94	232	64	72%	62	73%	84	64%	83	64%	72	69%	95	59%
1/31/94	137	64	53%	68	50%	75	45%	69	50%	57	59%	61	55%
2/14/94	95	70	27%	64	33%	57	40%	70	26%	104	-10%	53	44%
2/28/94	182	38	79%	45	75%	34	81%	42	77%	41	77%	32	83%
3/14/94	91	47	48%	49	46%	58	37%	75	18%	58	36%	55	40%
4/8/94	313	56	82%	62	80%	111	65%	99	68%	120	62%	72	77%
4/20/94	88	68	23%	67	24%	70	21%	67	24%	69	22%	58	34%
5/18/94	417	---	---	93	78%	105	75%	83	80%	---	---	---	---
6/9/94	399	---	---	125	69%	134	66%	162	59%	---	---	---	---
6/15/94	198	---	---	131	34%	159	20%	137	31%	---	---	---	---
7/21/94	215	109	49%	117	46%	128	40%	127	41%	115	47%	---	---
9/12/94	150	123	18%	115	23%	113	25%	96	36%	79	47%	96	36%
9/29/94	164	114	30%	145	12%	128	22%	153	7%	94	43%	94	43%
10/13/94	272	166	39%	178	35%	165	39%	208	24%	183	33%	107	61%
10/27/94	344	244	29%	275	20%	246	28%	240	30%	282	18%	225	35%
12/1/94	108	129	-19%	52	52%	88	19%	120	-11%	136	-25%	121	-12%
12/14/94	182	156	14%	149	18%	130	29%	136	25%	154	15%	156	15%
1/11/95	147	127	13%	56	62%	76	48%	98	33%	88	40%	109	26%
2/16/95	180	141	22%	114	37%	109	39%	158	12%	144	20%	142	21%
10/4/96	16	8	46%	9	41%	9	41%	6	63%	8	51%	---	---
10/18/96	20	7	68%	18	14%	19	7%	12	41%	12	42%	9	58%
10/25/96	24	15	37%	22	7%	22	7%	12	48%	15	37%	14	42%
11/1/96	34	24	32%	27	22%	28	19%	20	41%	22	37%	20	43%
Average:	148	73	41%	75	42%	78	42%	82	41%	80	33%	67	46%
Std. Dev.:	112	60	24%	59	23%	57	23%	62	22%	62	26%	54	22%
n:	30	27	27	30	30	30	30	30	30	27	27	25	25

Table 5.18. Total Kjeldahl nitrogen rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n^a	k_{20}	θ	R^2	C^{*c}	k_{20}	θ	R^2
Pond 4	0.60	0.09	0.99	0.63	10	10	1.01	0.74
Pond 5	0.60	0.11	0.99	0.58	10	12	1.00	0.51
Pond 6	0.60	0.10	0.98	0.68	10	12	1.01	0.62
Pond 7	0.60	0.10	0.99^b	0.61	10	11	1.01^d	0.48
Pond 8	0.60	0.08	0.99	0.73	10	8.2	1.01	0.84
Pond 9	0.60	0.11	0.99^b	0.67	10	10	1.01^d	0.71
Average =	0.60	0.10	0.99	0.65	10	10	1.01	0.65
Std. Dev. =	0.00	0.01	0.00	0.05	0.0	1.4	0.00	0.14

[a] n held constant at 0.6

[b] θ held constant at 0.99

[c] C^* held constant at 10 mg/l

[d] θ held constant at 1.01

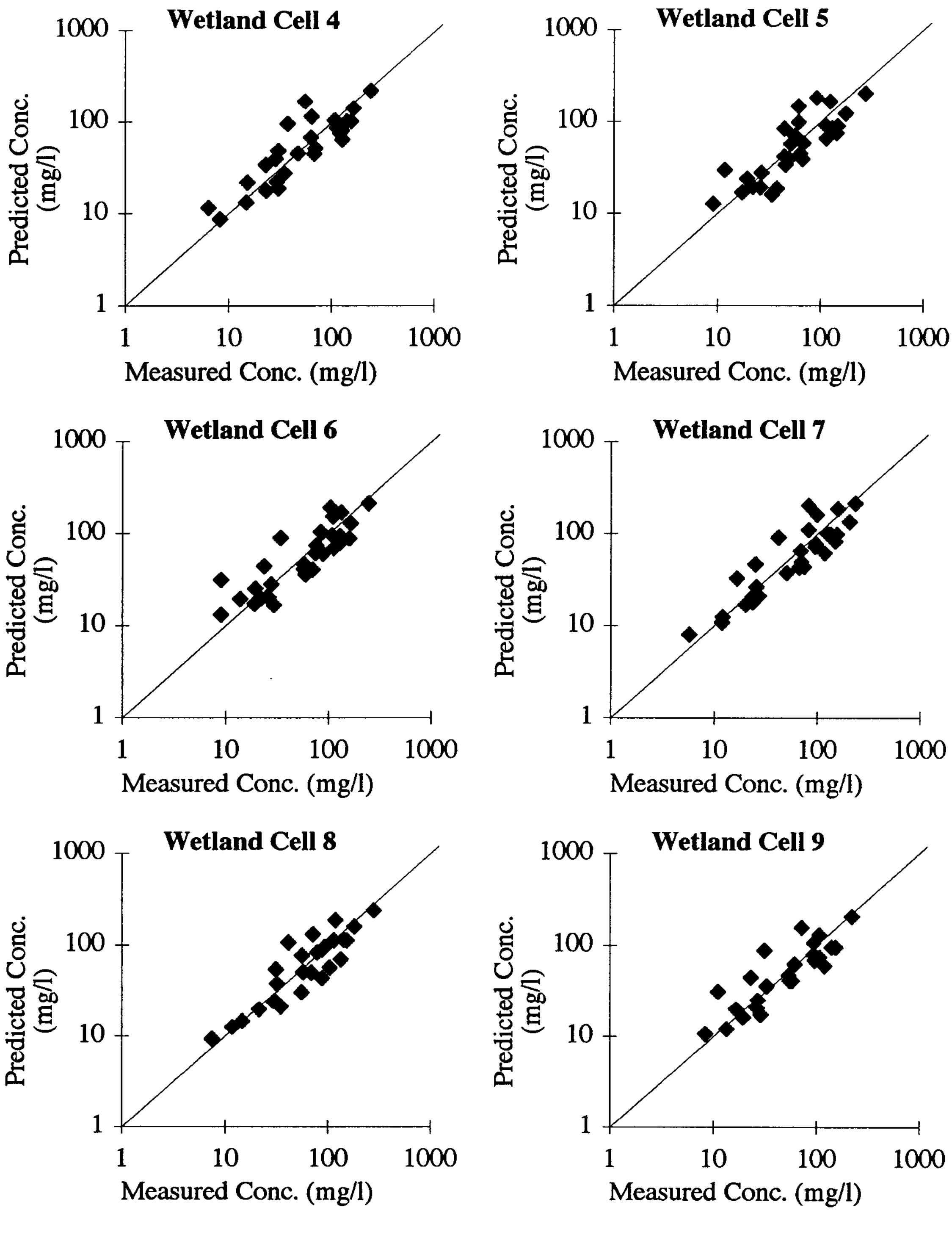

Figure 5.13. Scatterplot of measured versus predicted total Kjeldahl nitrogen concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. n held constant at 0.6.

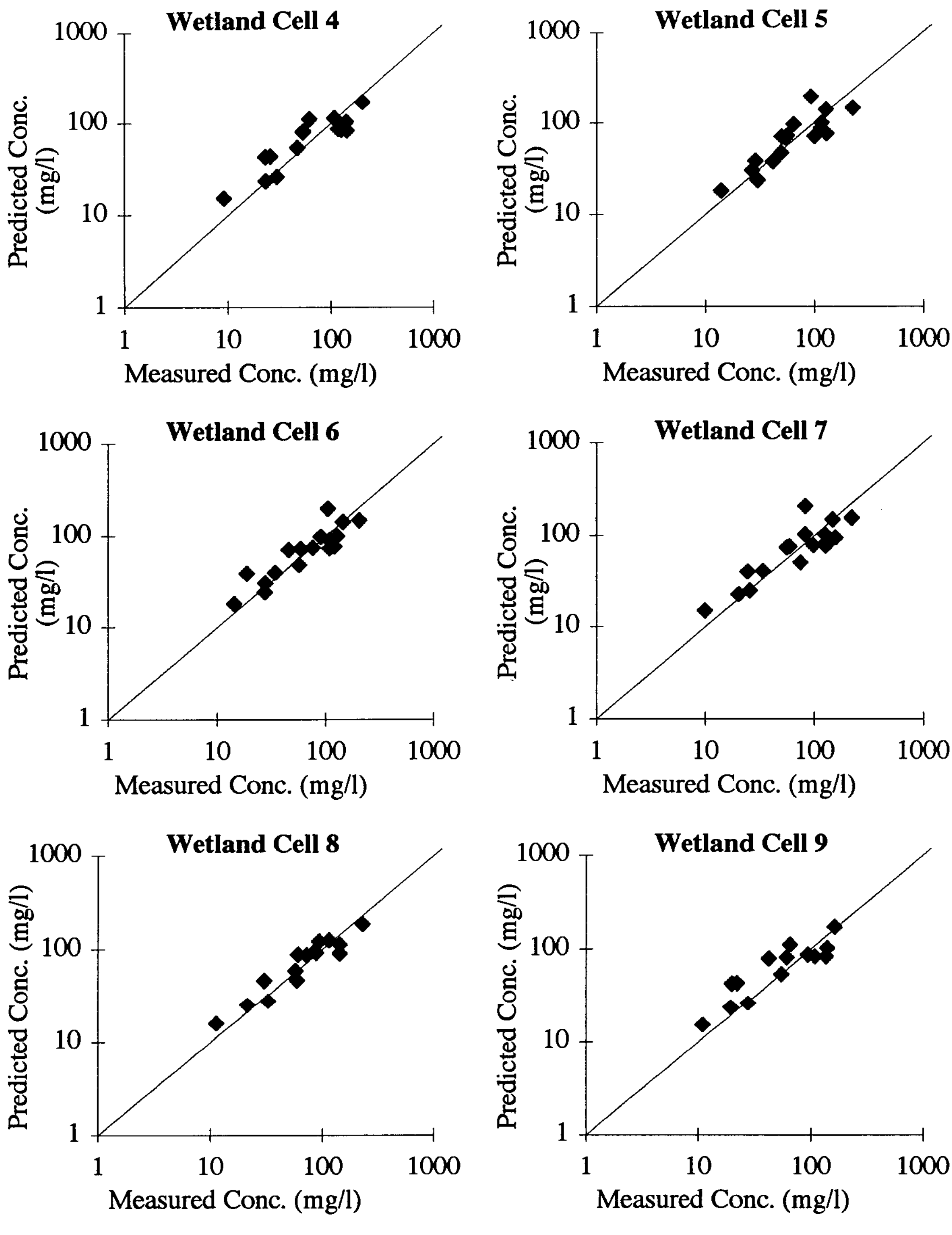

Figure 5.14. Scatterplot of measured versus predicted total Kjeldahl nitrogen concentrations using the k-C* model with C* held at 10 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.19. Ammonia concentrations (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	19	7	61%	18	3%	5	75%	12	37%	14	26%	8	60%
10/27/93	19	15	20%	21	-10%	11	43%	12	34%	14	27%	11	44%
11/4/93	16	14	15%	16	1%	12	23%	9	45%	15	7%	12	27%
11/17/93	12	12	0%	13	-6%	9	24%	8	32%	14	-15%	11	14%
12/2/93	64	19	70%	9	85%	4	94%	11	82%	28	56%	8	88%
12/16/93	68	25	63%	34	50%	47	31%	40	41%	80	-17%	29	57%
1/10/94	31	20	35%	10	69%	10	68%	15	51%	35	-14%	15	50%
1/17/94	151	42	72%	40	74%	57	63%	60	60%	47	69%	60	60%
1/31/94	121	59	51%	61	50%	72	40%	67	44%	59	51%	58	52%
2/14/94	90	62	31%	58	36%	55	39%	68	24%	95	-5%	50	44%
2/28/94	118	31	74%	38	68%	29	76%	34	71%	33	72%	27	77%
3/14/94	50	33	34%	33	34%	41	18%	53	-6%	40	20%	39	22%
4/8/94	215	42	81%	48	77%	76	65%	82	62%	90	58%	58	73%
4/20/94	162	61	63%	41	74%	70	57%	66	60%	66	59%	61	62%
5/4/94	240	---	---	47	81%	65	73%	62	74%	---	---	---	---
5/18/94	301	---	---	65	79%	83	73%	67	78%	---	---	---	---
6/9/94	293	---	---	108	63%	124	58%	126	57%	---	---	---	---
6/15/94	154	---	---	123	20%	137	11%	123	20%	---	---	---	---
7/21/94	134	93	31%	101	25%	96	28%	89	34%	76	44%	---	---
9/12/94	118	101	14%	89	25%	94	21%	88	26%	59	50%	83	30%
9/29/94	105	85	19%	107	-2%	104	1%	114	-9%	66	37%	71	33%
10/13/94	191	123	36%	135	29%	128	33%	151	21%	124	35%	81	58%
10/27/94	212	177	17%	194	8%	167	21%	161	24%	189	11%	146	31%
12/1/94	98	79	20%	32	68%	56	43%	72	26%	85	13%	72	26%
12/14/94	142	118	17%	124	13%	112	21%	110	23%	124	13%	117	18%
1/11/95	116	108	7%	57	51%	85	27%	98	16%	85	27%	104	11%
2/16/95	133	111	17%	110	17%	113	15%	113	15%	103	23%	95	28%
7/27/95	10	4	57%	5	56%	10	0%	5	52%	1	89%	7	35%
8/16/95	12	10	13%	11	8%	13	-10%	9	26%	---		9	22%
10/10/95	18	10	43%	23	-29%	17	5%	9	47%	8	53%	11	39%
11/21/95	18	14	23%	16	13%	18	-1%	14	23%	14	20%	13	25%
12/11/95	15	11	30%	12	22%	13	17%	10	35%	13	13%	10	32%
2/1/96	23	18	23%	19	17%	24	-3%	9	61%	22	4%	19	17%
10/4/96	8	3	65%	4	50%	5	32%	2	79%	3	59%	1	82%
10/11/96	7	7	2%	5	35%	6	18%	1	85%	5	39%	2	76%
10/18/96	10	3	74%	10	2%	11	-11%	5	54%	6	37%	3	67%
10/25/96	14	9	35%	12	11%	15	-6%	6	55%	10	30%	8	45%
11/1/96	24	16	35%	18	25%	17	28%	11	52%	16	34%	11	53%
Average:	93	45	37%	49	34%	53	32%	52	42%	50	31%	40	44%
Std. Dev.:	84	45	23%	46	30%	46	27%	47	23%	45	26%	39	21%
n:	38	34	34	38	38	38	38	38	38	33	33	33	33

Table 5.20. Ammonia rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n[a]	k_{20}	θ	R^2	C^{*}[c]	k_{20}	θ	R^2
Pond 4	0.60	0.09	1.00[b]	0.71	3.0	11	1.05[d]	0.75
Pond 5	0.60	0.13	1.00	0.55	3.0	13	1.02	0.56
Pond 6	0.60	0.10	0.99	0.68	3.0	12	1.02	0.71
Pond 7	0.60	0.16	1.04	0.67	3.0	16	1.06	0.69
Pond 8	0.60	0.09	1.01	0.80	3.0	10	1.05[d]	0.88
Pond 9	0.60	0.20	1.05	0.79	3.0	24	1.09	0.80
Average =	0.60	0.13	1.02	0.70	3.0	14	1.05	0.73
Std. Dev. =	0.00	0.04	0.02	0.09	0.0	5.4	0.02	0.11

[1] n held constant at 0.6

[2] θ held constant at 1.00

[2] C^{*} held constant at 3 mg/l

[4] θ held constant at 1.05

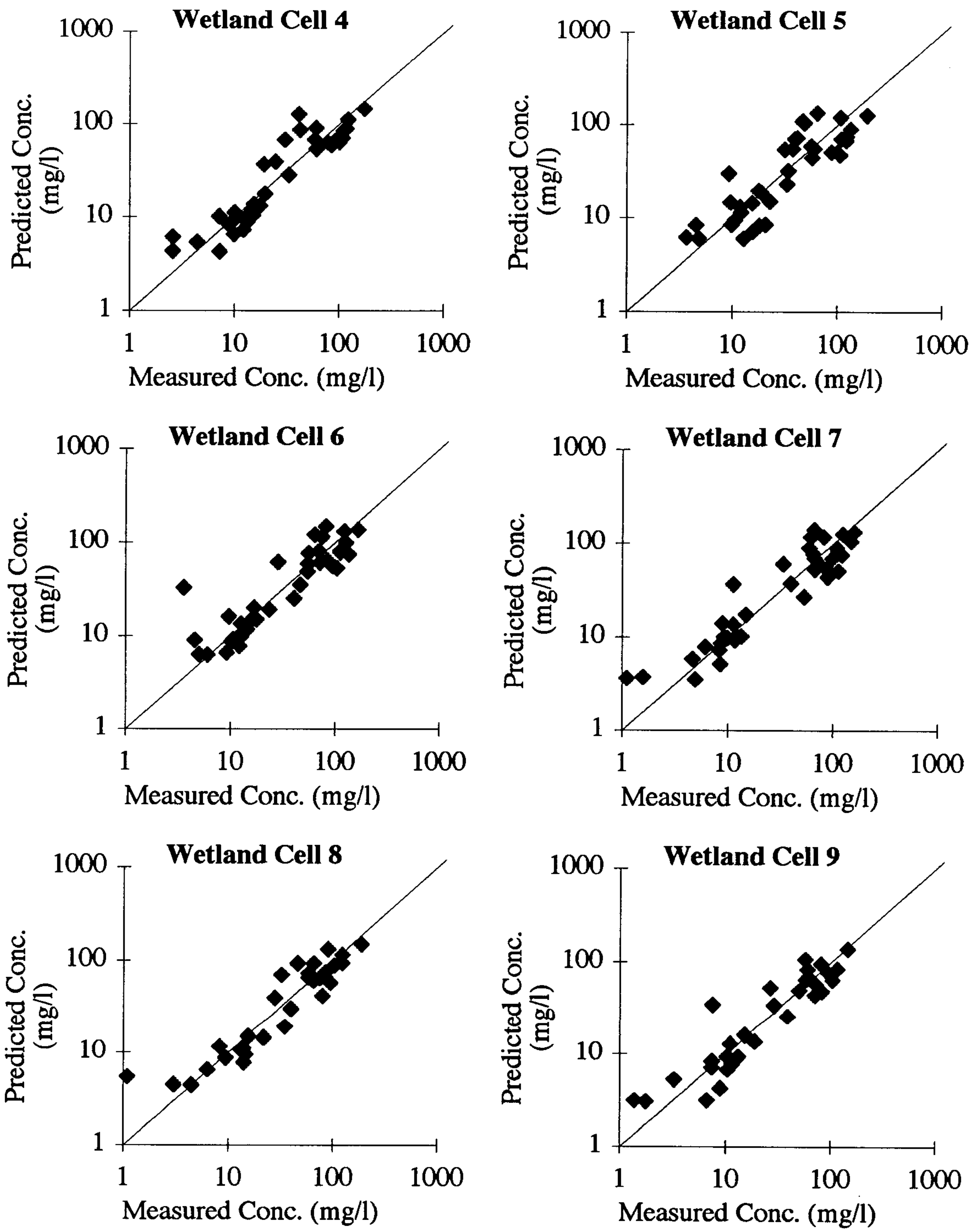

Figure 5.15. Scatterplot of measured versus predicted ammonia concentrations using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. n held constant at 0.6.

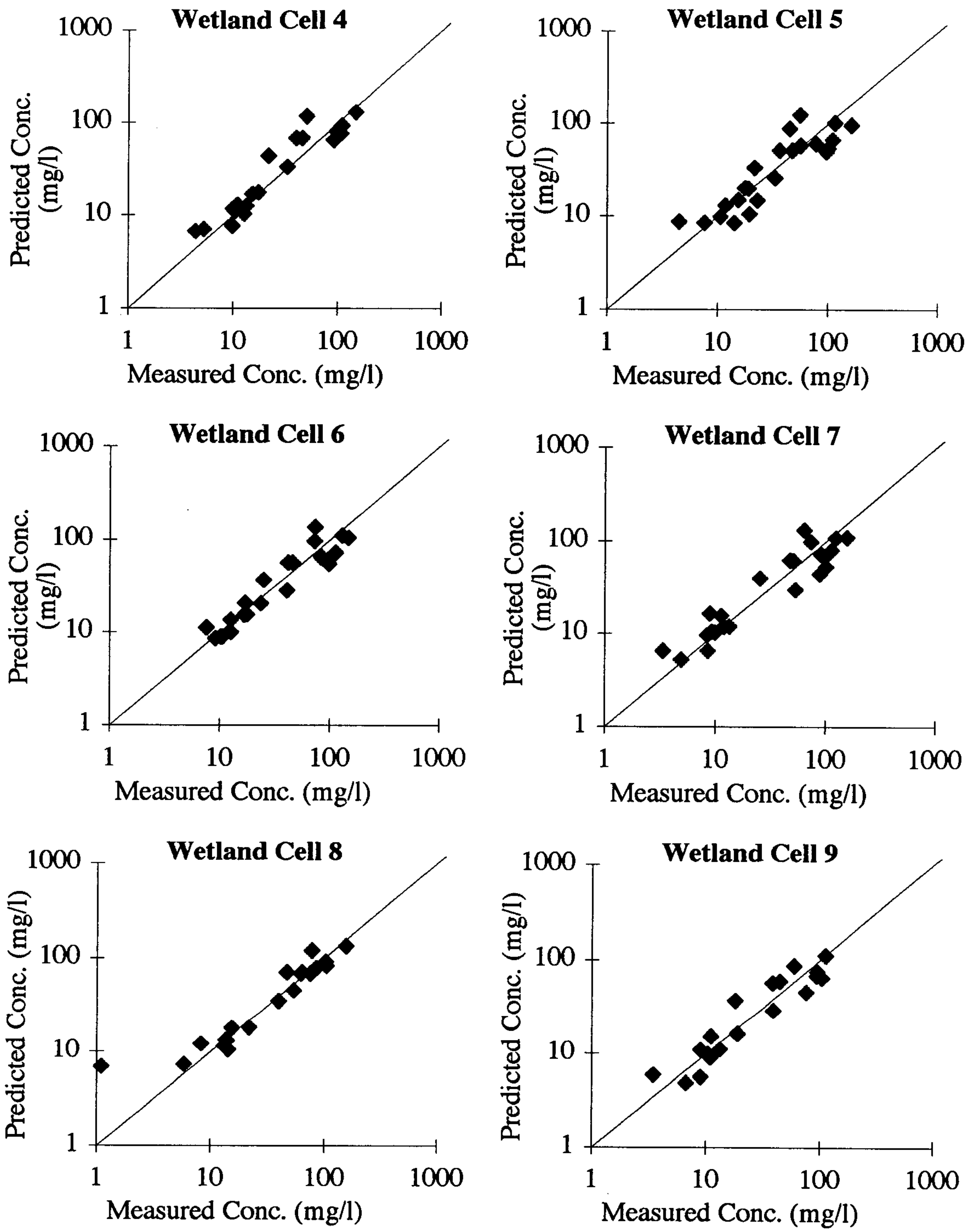

Figure 5.16. Scatterplot of measured versus predicted ammonia concentrations using the k-C^* model with C^* held at 3 mg/l for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.21. Nitrate concentrations (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	0.10	0.10	0%	0.10	0%	0.10	0%	0.10	0%	0.10	0%	0.10	0%
11/17/93	0.00	0.00	0%	0.00	0%	0.00	0%	0.00	0%	0.00	0%	0.00	0%
12/2/93	0.00	0.00	0%	0.00	0%	0.00	0%	0.00	0%	0.00	0%	0.00	0%
1/10/94	0.12	0.07	42%	0.05	58%	0.05	58%	0.30	-153%	0.17	-42%	0.06	50%
1/17/94	0.66	0.14	79%	0.13	80%	0.14	79%	0.16	76%	0.16	76%	0.17	74%
Average:	0.18	0.06	24%	0.06	28%	0.06	27%	0.11	-15%	0.09	7%	0.07	25%
Std. Dev.:	0.28	0.06	36%	0.06	39%	0.06	38%	0.13	83%	0.08	43%	0.07	35%
n:	5	5	5	5	5	5	5	5	5	5	5	5	5

5.6.11 Fecal Coliforms

Fecal coliforms were measured on 20 sampling dates during the high loading period. The average inlet concentration was 1,230,000 CFU/100 ml with a range from 43,000 to 5,860,000 CFU/100 ml (Table 5.22). Coliforms were reduced by an average of 80% with a range of 77 to 81%. First-order rate constants were fit to the data and are summarized in Table 5.23 and plots of the data are shown in Fig. 5.17 and 5.18. The fecal coliform loadings were too high to allow for an estimate of C^* to be found, so it was held constant at 10 CFU/100 ml for all wetland cells.

5.6.12 Dissolved Oxygen

The average DO, based on 27 sampling dates, was 3.3 mg/l with a range from 0.6 to 8.9 mg/l (Table 5.24). Outlet DO concentrations were almost always less than inlet DO concentrations. The average outlet DO concentration was 0.5 mg/l with a range of 0.3 to 0.7 mg/l.

5.6.13 pH

pH measurements were taken on 33 sampling dates. The average inlet pH was 7.30 with a range from 7.03 to 8.1 (Table 5.25). The average outlet pH was 7.05 with a range from 7.02 to 7.11. The outlet pH was generally slightly lower than inlet pH.

5.6.14 Conductivity

Conductivity was measured on 26 sampling dates, during the high loading period. The average inlet conductivity was 2,280 μmho/cm, with a range from 777 to 4,500 μmho/cm (Table 5.26). Outlet conductivity was an average of 21% lower.

Table 5.22. Fecal coliform counts (CFU/100 ml) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
1/17/94	175,500	22,000	87%	---	---	---	---	---	---	---	---	---	---
1/31/94	156,500	34,000	78%	32,000	80%	11,000	93%	20,000	87%	15,000	90%	29,000	81%
2/7/94	340,000	20,400	94%	21,000	94%	11,900	97%	17,800	95%	9,600	97%	12,000	96%
2/14/94	740,000	36,000	95%	40,500	95%	20,450	97%	46,500	94%	50,500	93%	28,500	96%
2/28/94	820,000	33,500	96%	38,500	95%	27,000	97%	34,000	96%	23,000	97%	22,000	97%
3/14/94	160,000	20,200	87%	16,650	90%	17,600	89%	29,100	82%	19,500	88%	25,150	84%
4/8/94	2,940,000	63,000	98%	157,000	95%	---	---	206,000	93%	223,000	92%	120,500	96%
4/20/94	1,500,000	134,000	91%	106,000	93%	247,000	84%	257,000	83%	228,000	85%	218,000	85%
5/4/94	680,000	---	---	5,200	99%	10,600	98%	12,500	98%	---	---	---	---
5/18/94	3,600,000	---	---	10,800	100%	16,400	100%	17,000	100%	---	---	---	---
6/9/94	248,000	---	---	12,100	95%	19,600	92%	18,600	93%	---	---	---	---
6/15/94	146,000	---	---	15,200	90%	16,700	89%	12,000	92%	---	---	---	---
7/21/94	43,000	9,400	78%	9,000	79%	13,864	68%	9,600	78%	9,300	78%	---	---
9/12/94	98,000	24,350	75%	31,600	68%	48,000	51%	9,200	91%	19,100	81%	---	---
9/29/94	97,375	31,325	68%	56,000	42%	50,000	49%	30,000	69%	34,000	65%	46,000	53%
10/13/94	470,000	47,000	90%	83,000	82%	42,700	91%	265,000	44%	80,000	83%	18,000	96%
10/27/94	155,000	149,000	4%	161,000	-4%	---	---	149,000	4%	213,000	-37%	135,000	13%
12/14/94	1,654,490	780,000	53%	720,000	56%	592,000	64%	645,000	61%	770,000	53%	665,000	60%
1/11/95	4,699,473	352,500	92%	58,000	99%	292,000	94%	384,500	92%	187,000	96%	308,500	93%
2/16/95	5,863,250	600,000	90%	630,000	89%	700,000	88%	840,000	86%	730,000	88%	700,000	88%
Average:	1,229,329	147,292	80%	115,976	81%	125,695	85%	158,042	81%	174,067	77%	179,050	80%
Std. Dev.:	1,702,262	230,711	24%	202,951	26%	213,268	16%	236,020	23%	248,588	34%	240,876	25%
n:	20	16	16	19	19	17	17	19	19	15	15	13	13

Table 5.23. Fecal coliform rate constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Wetland Cell	n^a	k_{20}	θ^b	R^2	C^{*3}	k_{20}	θ^b	R^2
Pond 4	0.60	0.49	1.01	0.51	10	36	1.01	0.46
Pond 5	0.60	0.52	1.01	0.39	10	40	1.01	0.29
Pond 6	0.60	0.48	1.01	0.66	10	36	1.01	0.61
Pond 7	0.60	0.44	1.01	0.66	10	34	1.01	0.64
Pond 8	0.60	0.46	1.01	0.50	10	34	1.01	0.43
Pond 9	0.60	0.47	1.01	0.60	10	35	1.01	0.56
Average =	0.60	0.48	1.01	0.55	10	36	1.01	0.50
Std. Dev. =	0.00	0.03	0.00	0.11	0.0	2.4	0.00	0.13

[a] n held constant at 0.6

[b] θ held constant at 1.01

[c] C^* held constant at 10 CFU/100ml

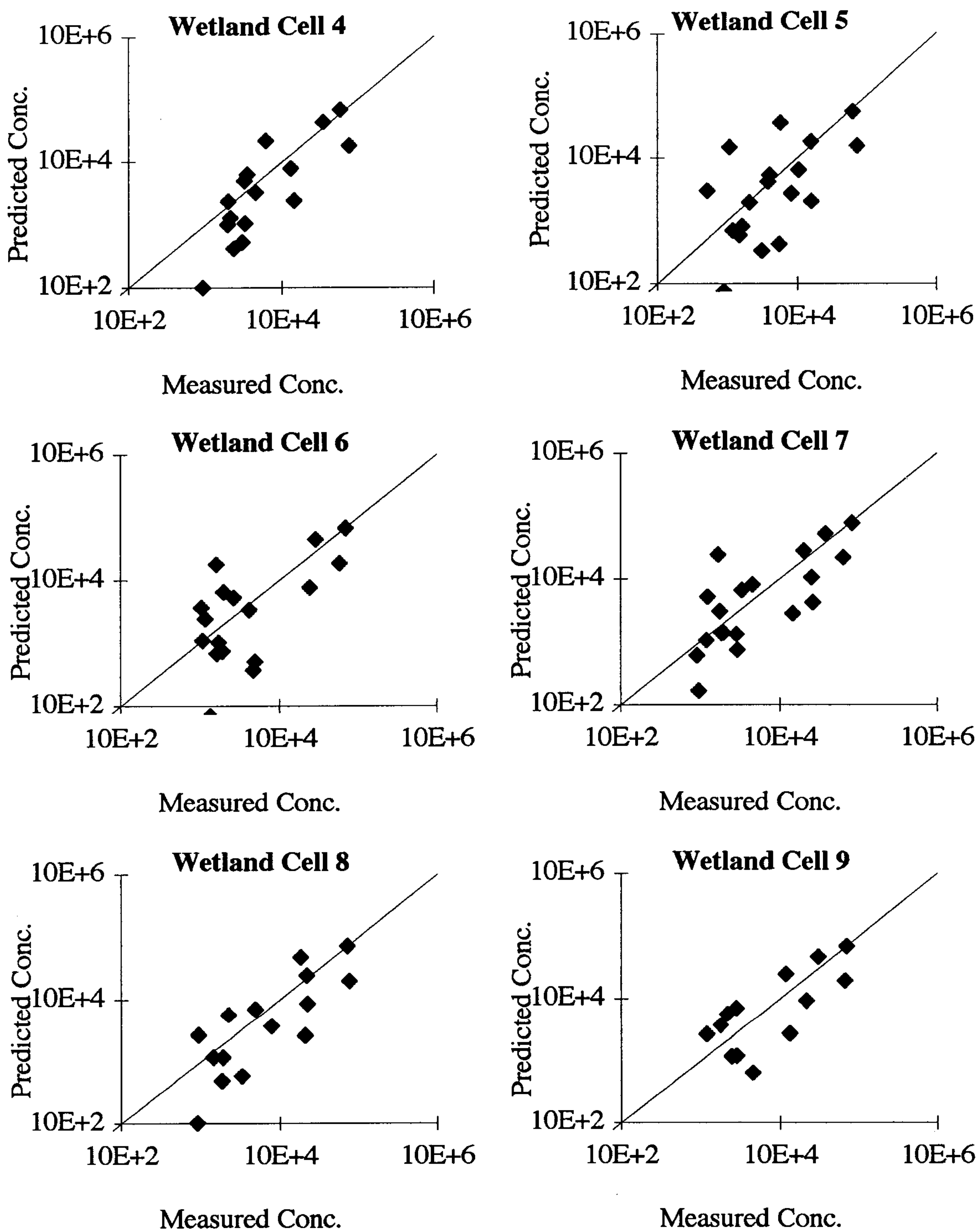

Figure 5.17. Scatterplot of measured versus predicted fecal coliform concentrations (CFU/100ml) using the volumetric model for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR. n and θ held constant at 0.6 and 1.01, respectively.

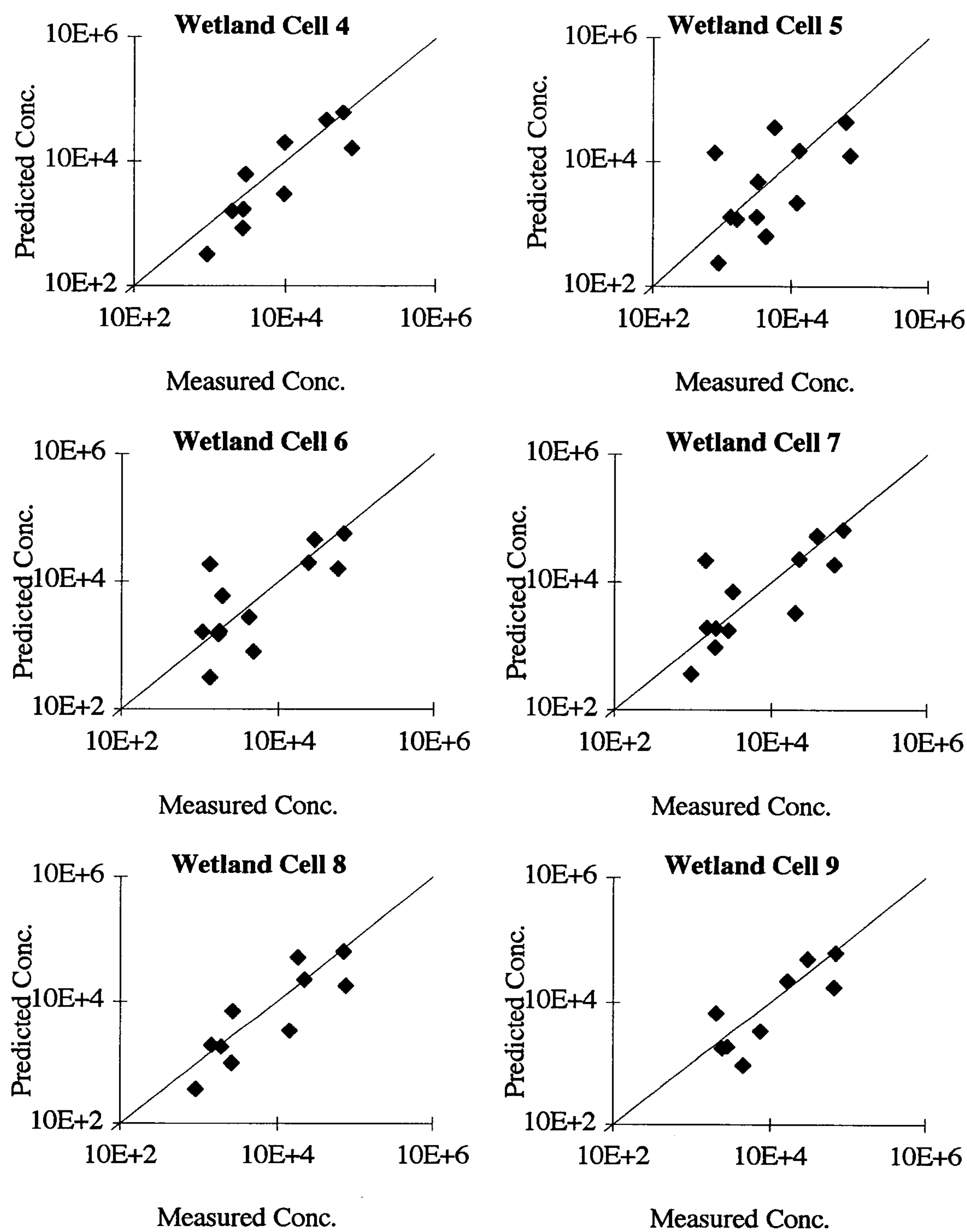

Figure 5.18. Scatterplot of measured versus predicted fecal coliform concentrations (CFU/100ml) using the k-C* model with C* held at 10 CFU/100ml and θ at 1.01 for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Table 5.24. Dissolved oxygen (mg/l) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/22/93	1.9	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%
10/27/93	6.4	0.1	98%	0.1	98%	0.1	98%	1.2	81%	0.1	98%	0.1	98%
11/4/93	8.2	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%
11/17/93	8.0	0.3	96%	0.3	96%	0.3	96%	0.3	96%	0.3	96%	0.3	96%
12/2/93	8.1	0.1	99%	0.1	99%	0.1	99%	0.8	90%	1.5	82%	0.1	99%
12/16/93	8.9	1.1	87%	0.9	90%	0.2	98%	0.4	95%	0.3	97%	0.3	97%
3/14/94	1.6	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.0	100%
4/8/94	1.0	0.3	75%	---	---	---	---	---	---	---	---	---	---
4/20/94	1.6	---	---	---	---	---	---	---	---	0.2	88%	0.0	100%
5/4/94	4.0	---	---	---	---	0.2	95%	1.0	75%	0.0	100%	0.0	100%
5/18/94	2.3	---	---	---	---	---	---	---	---	---	---	---	---
9/12/94	1.9	0.0	100%	0.0	100%	0.0	100%	0.0	100%	0.2	89%	0.2	89%
12/1/94	2.4	0.1	98%	---	---	0.1	96%	---	---	0.1	96%	0.3	88%
12/14/94	3.7	---	---	---	---	---	---	---	---	---	---	1.6	56%
1/11/95	1.5	0.1	93%	0.3	81%	0.1	93%	---	---	---	---	---	---
2/16/95	4.0	0.5	88%	---	---	---	---	---	---	---	---	---	---
7/27/95	0.8	0.1	87%	0.2	71%	0.1	89%	0.1	91%	1.0	-19%	0.0	100%
8/16/95	0.7	0.0	100%	0.4	53%	0.0	100%	0.2	73%	2.5	-235%	0.3	57%
10/10/95	0.1	0.0	77%	0.0	92%	0.0	100%	0.1	46%	---	---	0.6	-395%
11/21/95	0.3	0.0	91%	0.0	100%	0.0	100%	0.3	6%	0.2	31%	2.8	-782%
12/11/95	0.5	0.3	47%	0.2	62%	0.0	100%	0.6	-15%	0.1	82%	0.1	76%
2/1/96	1.1	0.8	29%	1.0	13%	0.6	51%	1.0	16%	0.8	33%	0.5	59%
10/4/96	3.2	2.0	38%	2.0	39%	1.5	53%	2.6	19%	3.3	0%	3.2	2%
10/11/96	0.6	0.5	17%	0.5	17%	0.4	44%	0.6	12%	1.0	-59%	1.1	-70%
10/18/96	8.0	2.9	64%	2.1	74%	2.1	74%	2.8	65%	2.9	64%	3.3	59%
10/25/96	6.1	0.2	97%	0.2	97%	0.1	98%	0.4	94%	0.2	97%	0.3	95%
11/1/96	1.7	0.1	93%	0.1	94%	0.1	95%	0.2	88%	0.2	87%	0.2	88%
Average:	3.3	0.4	81%	0.4	79%	0.3	90%	0.6	67%	0.7	54%	0.7	18%
Std. Dev.:	2.9	0.7	25%	0.6	28%	0.5	18%	0.8	38%	1.0	80%	1.0	204%
n:	27	23	23	20	20	22	22	20	20	21	21	23	23

Table 5.25. pH for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 5	Cell 6	Cell 7	Cell 8	Cell 9
10/27/93	7.37	7.38	7.46	7.42	7.47	7.52	7.57
11/4/93	7.35	7.19	7.23	7.02	7.08	7.10	7.26
11/17/93	7.04	6.79	6.88	6.74	6.75	6.91	6.85
12/2/93	7.37	7.04	6.82	6.66	6.64	6.93	6.68
12/16/93	7.35	7.18	6.98	7.08	7.07	7.11	6.90
1/10/94	7.51	7.24	6.95	6.74	6.76	6.92	6.93
1/17/94	7.77	7.24	7.12	6.98	7.04	6.89	7.03
1/31/94	7.54	7.21	7.02	6.78	7.02	7.01	7.12
2/14/94	7.29	7.04	6.90	6.83	6.94	6.80	6.98
2/28/94	7.75	7.05	7.20	6.84	7.07	7.16	7.07
3/14/94	7.26	6.70	6.57	6.54	6.61	6.67	6.70
4/8/94	7.17	6.58	6.49	6.28	6.56	6.55	6.62
4/20/94	7.18	6.67	6.61	6.48	6.57	6.63	6.70
7/21/94	7.44	7.34	7.24	7.10	7.51	7.50	- - -
9/12/94	7.39	7.06	6.94	6.94	7.26	6.93	7.56
9/29/94	7.61	7.47	7.19	7.13	7.22	7.06	7.66
10/13/94	7.63	7.25	7.23	7.39	7.31	7.53	7.57
10/27/94	7.51	7.56	7.72	7.58	7.61	7.63	7.63
12/1/94	7.76	7.65	7.31	7.44	7.48	7.54	7.62
12/14/94	7.91	7.48	7.41	7.76	7.33	7.48	7.93
1/11/95	7.23	7.16	6.83	6.99	7.00	7.00	7.14
2/16/95	8.10	7.93	7.83	7.83	7.77	7.91	7.92
7/27/95	6.73	7.03	6.83	6.97	6.52	6.95	7.01
8/16/95	7.04	6.93	7.08	6.93	7.07	6.86	7.18
10/10/95	7.03	6.82	7.21	6.94	6.84	6.83	6.96
11/21/95	7.05	6.90	7.26	7.20	7.06	7.08	6.98
12/11/95	6.82	6.82	6.91	6.97	6.87	6.88	6.88
2/1/96	7.14	7.08	7.09	7.25	6.95	7.03	7.01
10/4/96	6.74	6.02	6.51	6.69	6.77	6.72	6.74
10/11/96	6.84	6.77	6.79	6.76	6.88	6.71	6.81
10/18/96	6.78	6.56	6.73	6.78	6.74	6.73	6.81
10/25/96	7.04	6.95	6.98	7.15	7.03	6.91	6.88
11/1/96	7.10	6.61	6.82	7.05	6.92	6.98	6.95
Average:	7.30	7.05	7.03	7.01	7.02	7.04	7.11
Std. Dev.:	0.35	0.37	0.31	0.35	0.32	0.32	0.37
n:	33	33	33	33	33	33	32

Table 5.26. Conductivity (umho/cm) for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

Date	Inlet	Cell 4	Cell 4 % Reduction	Cell 5	Cell 5 % Reduction	Cell 6	Cell 6 % Reduction	Cell 7	Cell 7 % Reduction	Cell 8	Cell 8 % Reduction	Cell 9	Cell 9 % Reduction
10/27/93	770	540	30%	680	12%	570	26%	610	21%	660	14%	570	26%
11/4/93	840	700	17%	800	5%	730	13%	620	26%	770	8%	730	13%
11/17/93	890	690	22%	730	18%	720	19%	630	29%	790	11%	710	20%
12/2/93	1,100	470	57%	380	65%	220	80%	390	65%	640	42%	360	67%
12/16/93	1,110	640	42%	830	25%	950	14%	770	31%	1,130	-2%	1,050	5%
1/10/94	800	650	19%	440	45%	420	48%	550	31%	1,100	-38%	1,060	-33%
1/17/94	1,400	1,100	21%	1,100	21%	1,130	19%	1,140	19%	1,100	21%	1,400	0%
1/31/94	1,200	1,130	6%	1,120	7%	1,140	5%	1,130	6%	1,120	7%	1,120	7%
2/14/94	1,140	1,130	1%	1,130	1%	1,120	2%	1,140	0%	1,180	-4%	1,110	3%
2/28/94	1,800	820	54%	950	47%	780	57%	860	52%	870	52%	760	58%
3/14/94	1,120	1,100	2%	1,100	2%	1,110	1%	1,120	0%	1,100	2%	1,100	2%
4/8/94	3,300	1,100	67%	1,200	64%	1,600	52%	1,700	48%	1,800	45%	1,200	64%
4/20/94	2,800	1,400	50%	1,200	57%	1,500	46%	1,500	46%	1,500	46%	1,400	50%
5/4/94	3,600	---	---	1,400	61%	1,600	56%	1,600	56%	---	---	---	---
5/18/94	4,400	---	---	1,800	59%	2,000	55%	1,800	59%	---	---	---	---
6/9/94	4,500	---	---	2,600	42%	2,700	40%	2,800	38%	---	---	---	---
6/15/94	2,900	---	---	2,600	10%	2,800	3%	2,600	10%	---	---	---	---
7/21/94	2,900	2,400	17%	2,400	17%	2,600	10%	2,500	14%	2,200	24%	---	---
9/12/94	2,700	2,400	11%	2,600	4%	2,600	4%	2,500	7%	2,200	19%	2,400	11%
9/29/94	2,800	2,600	7%	2,800	0%	2,600	7%	2,900	-4%	2,600	7%	2,500	11%
10/13/94	3,900	3,400	13%	3,200	18%	3,500	10%	3,300	15%	2,700	31%	2,800	28%
10/27/94	4,300	3,800	12%	4,200	2%	4,000	7%	3,800	12%	4,200	2%	3,600	16%
12/1/94	2,251	1,900	16%	1,000	56%	1,600	29%	2,100	7%	2,400	-7%	2,000	11%
12/14/94	2,165	2,400	-11%	2,400	-11%	2,200	-2%	2,200	-2%	2,600	-20%	2,300	-6%
1/11/95	2,062	2,200	-7%	1,200	42%	1,800	13%	2,000	3%	1,900	8%	2,100	-2%
2/16/95	2,495	2,200	12%	2,200	12%	2,400	4%	2,300	8%	2,300	8%	2,200	12%
Average:	2,279	1,580	21%	1,618	26%	1,707	24%	1,714	23%	1,675	13%	1,546	17%
Std. Dev.:	1,209	964	21%	967	24%	976	23%	939	21%	897	22%	851	25%
n:	26	22	22	26	26	26	26	26	26	22	22	21	21

5.7 Discussion

The OSUDWTS was loaded at both high and low mass loading rates during the study period. This gave a data set with a range of concentrations, which allowed the design equations to be checked at both high and low loadings.

5.7.1 Temperature

Generally wetland water temperature is assumed to be close to mean daily air temperature (Kadlec and Knight, 1996). This was not the case at the OSUDWTS. The water temperature in the wetlands was found to be linearly related to air temperature (Eq. 5-10), however, the relationship was not one-to-one. It is hypothesized that the wetland water temperature lagged behind the mean daily air temperature and that the average of the mean daily air temperatures for the previous few days may result in a one-to-one relationship. Equation 5-10 was used to calculate daily wetland water temperature for all sampling days where temperature data was not collected.

5.7.2 Chemical Oxygen Demand

COD is a measurement of the total amount of oxygen required to completely oxidized all organic matter and chemical constituents in a waste sample. COD is not generally measured because much of the material is recalcitrant and will not be oxidized naturally. In a constructed wetland loaded with livestock wastewater, COD is generally much higher than BOD. Little data exist for COD in constructed wetlands and wetland systems are rarely designed based on COD. However, COD measure can give an indication of long-term oxygen demand and may be important in livestock wastewaters. Of the 68 wetlands in the LWDB, COD measurements were only taken at two study sites: Auburn Poultry and the OSUDWTS (Knight et al., 1996 draft). The average COD loading at the OSUDWTS was 815 kg/ha-d and the average inlet concentration was 2,181 mg/l.

The loading at the OSUDWTS was very high, 10 to 20 times greater than the loading to constructed wetlands treating domestic wastewater (Kadlec and Knight, 1996). However, the wetlands still removed an average of 45% of the COD.

Temperature had no effect on the rate constants in the volumetric model ($\bar{\theta}$ = 1.0) and a significant effect in the areal model ($\bar{\theta}$ = 1.12) (Table 5.7). However, both models provided an equal fit to the data, and appeared to be adequate for predicting COD transformations.

5.7.3 Biochemical Oxygen Demand

BOD is a measurement of the oxygen consumption by microorganisms during the oxidation of organic matter and inorganic materials (Kadlec and Knight, 1996). The test is usually run for 5 days (BOD_5) and indicates the amount of readily degradable organic matter. BOD is the most common wastewater parameter used for design and evaluation of constructed wetlands.

Typical loading rates to constructed wetlands treating domestic wastewaters are less than 100 kg/ha-d with an inlet concentration between 10 - 100 mg/l (Kadlec and Knight, 1996). Average loading at the OSUDWTS, 188 kg/ha-d, was approximately twice the typical loading rate for domestic wastewater wetlands. The inlet concentrations for the OSUDWTS ranged from 39 to over 2,000 mg/l (with an average of 502 mg/l). The average inlet BOD for dairies from the LWDB was 404 mg/l (Knight et al., 1996 draft). The overall average removal of BOD's for the OSUDWTS was 52% which was slightly lower than the 68% average removal reported for dairy wetlands (Knight et al., 1996 draft).

Temperature had an effect on the rate constants in both the volumetric ($\bar{\theta}$ = 1.02) and areal ($\bar{\theta}$ = 1.07) models. The volumetric rate constant for the OSUDWTS ($\bar{k}_{20}$ = 0.23 d^{-1}) was much lower than the rate constant reported for domestic wastewater ($\bar{k}_{20}$ =

$0.678\ d^{-1}$) (Reed, 1995 draft). This may be the result of the extremely high loading rates, which caused low DO levels, which, in turn, decreased removal of BOD. Theta for the volumetric model ($\bar{\theta}$ = 1.02) was also lower than the reported theta for domestic wastewater wetlands (θ = 1.06) (Reed, 1995 draft). The general fit of the model appears to be correct (Fig. 5.5) but considerable variability still exists ($\overline{R^2}$ = 0.63).

The average areal rate constant ($\bar{k}_{20}$ = 29 m/yr) was slightly higher than the rate constant reported for constructed wetlands treating livestock waste ($\bar{k}_{20}$ = 22 m/yr) (Knight et al., 1996 draft) but much lower than the $\bar{k}_{20}$ reported for constructed wetlands treating domestic wastewater ($\bar{k}_{20}$ = 34 m/yr) (Kadlec and Knight, 1996). As in the volumetric model, oxygen limitations may cause the lower rate constants. The average areal theta for the OSUDWTS was found to be 1.07, which suggests a fairly strong temperature dependence. A $\bar{\theta}$ of 1.03 is reported for wetlands in the LWDB (Knight et al., 1996 draft) and data for domestic wastewaters indicate no temperature dependence in the k-C^* model (Kadlec and Knight, 1996).

Both models explained an equal amount of the variability for all cells ($\overline{R^2}$ = 0.63). However, the standard deviation associated with the R^2's for the volumetric model was less than the standard deviation associated with the R^2's for the areal model.

5.7.4 Total Solids

TS is a measurement of both dissolved and suspended solids. Suspended solids removal in wetlands is a function of the wastewater and wetland characteristics. Several factors including particle density, particle size, water velocity, water depth, and the shape of the particle effect suspended solids removal. If all of these factors are known then the settling velocity can be determined and the time and distance of travel can be calculated.

However, these data are rarely available or measurable (Kadlec and Knight, 1996). It has been shown that TSS decrease exponentially in wetlands and that a first-order model is a good representation of TS removal (Kadlec and Knight, 1996). It was assumed that TS could also be modeled using the first-order model.

TS data are rarely collected for constructed wetlands and no design equation exists to predict its removal. However, a measurement of TS may be important if a wastewater contains a high percentage of dissolved solids. This was found to be the case for the OSUDWTS where TS concentrations were more than three times TSS concentrations.

Removal of TS averaged 27% with a range of 15 to 31%. This low removal indicates that the OSUDWTS was not very efficient at removing dissolved solids, which has shown to be the case in most constructed wetlands (Kadlec and Knight, 1996). Both the volumetric and areal models adequately predicted removal of TS ($\overline{R}^2$ = 0.72 and 0.75, respectively). The volumetric model showed no dependence on temperature (θ = 1.00) while the areal model showed a slight temperature dependence (θ = 1.03). However, the rate constants for both the volumetric and areal models varied widely (0.06 - 0.12 d^{-1} and 6.35 - 13.18 m/yr, respectively), and provided little information for design purposes.

5.7.5 Total Suspended Solids

Removal of suspended solids is a major function performed by wetlands. It is one of the most common parameters measured in wetlands and the design of wetlands systems is often based on TSS removal. Settling is the major mechanism for removal of TSS and is based on settling velocities. Wetlands also produce dissolved and suspended solids. Microorganisms, plant material, litter, and re-suspension of settled solids all contribute to the TSS in a wetland. As mentioned in the previous section, first-order models have been shown to be accurate for predicting TSS removal.

The average inlet concentration (542 mg/l) at the OSUDWTS was less than the average inlet concentration (914 mg/l) reported for diary wetlands in the LWDB and may be a result of the pretreatment at OSUDWTS (Knight et al., 1996 draft). Recall, that the wastewater at OSUDWTS passes over a solids separator before being loaded to the wetland cells. The average reduction for TSS dairy wetlands in the LWDB was 53%, which was very close to the average reduction of 55% found for the OSUDWTS. However, removal rates of TSS were highly variable in both OSUDWTS and the LWDB (Knight et al., 1996 draft).

Volumetric and areal rate constants that were fitted to the data explained almost none of the variability (R^2 = 0.03 and 0.01, respectively). Temperature was assumed to have a slight effect on treatment (Kadlec and Knight, 1996) and theta values in both models were held constant at 1.01. These models are inadequate to explain TSS removal and should not be used. In addition, removal of TSS was much lower than reported for wetlands treating domestic wastewater, in which TSS are reduced to near background levels regardless of loadings. Kadlec and Knight (1996) report TSS removal can be predicted using:

$$\text{“}C^* = C_o = 5.1 + 016 \cdot C_i \qquad (5\text{-}11)$$

where, C^* = background concentration (mg/l),

C_o = effluent concentration (mg/l),

C_i = influent concentration (mg/l),

R^2 = 0.23; N = 1,582

standard error in C_o = 15

$0.1 < C_i < 807$ mg/l

$0.0 < C_o < 290$ mg/l.”

Using this equation and the OSUDWTS' average inlet concentration of 542 mg/l, the outlet concentration would be predicted to be 92 mg/l. The actual average outlet concentration was 142 mg/l. Therefore, this equation also fails to accurately predict TSS removal for the pretreated wastewater at OSUDWTS.

5.7.6 Total Phosphorus

Long-term phosphorus removal in wetlands is primarily a result of sedimentation of particulate phosphorus and sorption of soluble phosphorus on soil particles. Plants and microbes also take up phosphorus but it is usually released back into the wetland. However, long-term storage can occur in undecomposed litter. Phosphorus removal in newly constructed wetlands is often high due to the availability of minerals such as aluminum or iron which can bind with phosphorus. However, the binding sites are quickly filled and the wetland can become phosphorus "saturated." Long-term removal rates of phosphorus in constructed wetlands are generally much lower than the removal rates of solids or BOD (Kadlec and Knight, 1996).

The average loading rate of phosphorus was 12 kg/ha-d and the average inlet concentration was 33 mg/l. The average removal rate was 42%. This was a fairly high removal rate and it is expected that as the system matures removal rates will decrease. Phosphorus removal in constructed wetlands has been represented with first-order models but rate constants are highly variable (Kadlec and Knight, 1996). The average areal rate constant for the OSUDWTS (k_{20} = 12 m/yr) was higher than the average rate constant reported in the LWDB (8 m/yr) (Knight et al., 1996 draft). Temperature had little effect on treatment in the volumetric model (θ = 0.99) and a slight effect on treatment in the areal model (θ = 1.02). Both models were fairly accurate at predicting outlet concentrations (Fig. 5.11 and 5.12) and had average R^2's of 0.61 and 0.72 for the volumetric and areal models, respectively. Both models appear to be adequate for predicting phosphorus removal but care must be taken if these equations are used to predict long-term removal.

5.7.7 Orthophosphate

PO_4-P is the common ionic form of phosphorus. PO_4-P was found to account for approximately 50% of the total phosphorus loading during the start-up period. An overall average of 43% of the PO_4-P was removed but the range of average removals was from 20 to 59%. The data set was too small to evaluate removal or design models.

5.7.8 Total Kjeldahl Nitrogen

TKN is a measurement of organic nitrogen and ammonia present in a sample. Nitrogen is generally the target waste constituent for removal in livestock wastewater wetlands. Nitrogen cycling in wetlands is very complex. Details of the nitrogen cycling processes are discussed in Chapter 6. It is generally believed that the primary pathway for nitrogen removal in constructed wetlands is via denitrification, which is the biological conversion of nitrate to nitrite to nitrogen gas under anaerobic conditions. Even though much is known about nitrogen cycling, current design equations are based on simple first-order models. This is due to the fact that most data available for constructed wetlands consist of only inlet and outlet data and a model with greater detail cannot be validated.

The TKN procedure does not account for any nitrate or nitrite that is present in the sample. Measurements of nitrate at the site were less than 1 mg/l and generally less than 0.12 mg/l (Table 5.21). Nitrite is rapidly converted to nitrate so it can be assumed to be zero. Therefore, for purposes of this discussion, the TKN measurement will be considered equal to the total nitrogen load in the water sample.

TKN mass loading to the OSUDWTS averaged 55 kg/ha-d with an average concentration of 148 mg/l (range = 16 - 417 mg/l). This was very close to the average TKN concentration reported in the LWDB (174.4 mg/l) (Knight et al., 1996 draft). The average removal for TKN was 41% for the OSUDWTS which is slightly lower the overall 49% removal reported in the LWDB (Knight et al., 1996 draft).

Volumetric and areal models did a fair job of predicting TKN removal ($\overline{R}^2$ = 0.65 for both models). However, the standard deviation associated with the R^2 was less for the volumetric model (σ = 0.05) than that for the areal model (σ = 0.14). This indicates that the volumetric model may predict TKN removal slightly better than the areal model. The rate constants developed for the areal model ($\overline{k}_{20}$ = 10 m/yr) are slightly lower than the $\overline{k}_{20}$ reported for wetlands in et LWDB ($\overline{k}_{20}$ = 14 m/yr) but it is within the range reported ($5 < \overline{k}_{20} < 32$ m/yr) (Knight et al., 1996 draft). Both models appear to be appropriate for predicting TKN removal at both high and low loadings (Fig. 5.13 and 5.14).

5.7.9 Ammonia

Constructed wetlands are often designed for removal of total ammonia. The major pathway for removal of ammonia in constructed wetlands is sequential nitrification-denitrification (see Chapter 6). Ammonia volatilization usually is not thought to be a significant pathway for loss of ammonia in wetlands treating domestic wastewater but it may be very important in wetlands treating livestock waste due to the high concentration of ammonia (Payne, 1996 draft) (see Chapter 6).

The average inlet and outlet concentrations reported in the LWDB for wetlands treating dairy waste are 74 mg/l and 30 mg/l, respectively (Knight et al., 1996 draft). The average inlet concentration at the OSUDWTS was 93 mg/l and the average overall reduction was 37%.

An average k_{20} of 10 m/yr and a θ of 1.05 was reported for wetlands in the LWDB (Knight et al., 1996 draft). At the OSUDWTS, the average k_{20} for the volumetric and areal models were 0.13 d^{-1} ($\sigma = 0.04$) and 14 m/yr ($\sigma = 5.37$), respectively. Average values of 1.02 ($\sigma = 0.02$) and 1.05 ($\sigma = 0.02$) were found for the volumetric and areal models, indicating a temperature dependence. Both models did an adequate job at predicting removal (Fig. 5.15 and 5.16) over the wide range of loadings (7 - 301 mg/l). The $\overline{R}^2$ for the volumetric model and areal models were 0.70 and 0.73, respectively.

5.7.10 Nitrate

Excess NO_3^- can lead to eutrophication of surface waters and can be a potential health hazard if found in drinking water. Fortunately, NO_3^- removal in wetlands is generally close to 100% (Kadlec and Knight, 1996). This is a result of sequential transformation of NO_3^- under anaerobic conditions to nitrogen gas (termed "denitrification"). Denitrification occurs very quickly and any NO_3^- present in a constructed wetland is usually rapidly denitrified which leads to NO_3^- measurements of near zero in wetlands. Details of denitrification are discussed in Chapter 6.

All measurements of inlet NO_3^- concentrations were below 1.00 mg/l. Outlet concentrations were always less than 0.17 mg/l and usually near zero which indicates that any NO_3^- present was denitrified.

5.7.11 Fecal Coliforms

Fecal coliforms are indicator organisms that are used to indicate the possible presence of the pathogenic organisms associated with human and animal waste. Pathogenic organisms are adapted to live in the host organism and, once out in the environment, their numbers are reduced as a result of natural die-off, predation, sedimentation, and an inability to adapt to higher or lower temperatures, ultraviolet exposure, and unfavorable water chemistry. Wetlands are good at removing pathogenic organisms, however, background concentrations rarely if ever reach zero due to the fecal inputs from wildlife (Kadlec and Knight, 1996). Background concentrations in constructed and natural wetlands range from 10 to 500 CFU/100 ml (Kadlec and Knight, 1996).

Wetlands treating livestock wastewaters can have very high inlet fecal coliform concentrations depending on the pretreatment. Average inlet concentration of fecal coliforms for the LWDB was 160,477 CFU/100 ml and effluent concentrations were 13,424 CFU/100 ml for an average reduction of 92%. The inlet concentration for the OSUDWTS was 1,230,000 CFU/100 ml and the average overall reduction was 81%. While these removal rates are high, the outlet concentrations are still high and the discharge to receiving waters would not be permitted without additional treatment.

Preliminary estimates indicate a k_t of 75 m/yr for surface flow wetlands treating domestic wastewater (Kadlec and Knight, 1996). The areal rate constant found for the OSUDWTS was 36 m/yr with a range from 34 to 40 m/yr. The volumetric and areal models adequately predicted fecal coliform removal ($\overline{R}^2$ = 0.55 and 0.56, respectively).

5.7.12 Dissolved Oxygen

DO is very important in most wastewater treatment processes such as nitrification and BOD reduction. Wetlands are unique because they generally have an aerobic water column and an anaerobic sediment layer. This allows for such processes as nitrification (aerobic process) and denitrification (anaerobic process) to occur. However, if wetlands are heavily loaded with high oxygen demand wastewaters, the available oxygen is quickly consumed and anaerobic conditions may exist throughout the water column. Many of the biological removal processes are much slower or completely inhibited by anaerobic conditions.

Only a few sites in the LWDB reported DO concentrations. For those that did report values, the average inlet DO was 2.53 mg/l and outlet concentrations were 1.57 mg/l. The average inlet and outlet DO for the OSUDWTS was 3.3 mg/l and 0.52 mg/l, respectively. The low DO at the OSUDWTS probably was responsible for slowing both the BOD_5 and NH_3 removal, and may explain why the rate constants were lower than the rate constants reported for domestic wastewaters.

5.7.13 pH

pH is a measurement of the hydrogen-ion concentration and is a way of expressing the alkalinity or acidity of a solution. Many treatment processes have been found to be influenced by pH, such as ammonia volatilization and phosphorus removal. Wastewaters are generally highly buffered, which results in a near neutral pH for constructed wetlands (Kadlec and Knight, 1996). The average inlet and outlet pH reported for wetlands in the LWDB were 7.54 and 7.50 respectively. An average inlet and outlet pH of 7.30 and 7.04 was found at the OSUDWTS (Table 5.25). This near neutral pH should have very little effect on most treatment processes.

5.7.14 Conductivity

Conductivity is a measurement of the ability of a solution to carry an electrical charge and is generally proportional to total dissolved solids or salinity. Most natural inland surface waters have a conductivity between 10 - 300 μmho/cm (Kadlec and Knight, 1996). Wetlands are thought to have little effect on conductivity except for dilution and concentration caused by rainfall and ET (Knight et al., 1996 draft). However, the average inlet and outlet conductivity for the OSUDWTS was 2,300 and 1,640 μmho/cm. The overall average reduction in conductivity was 31% and it did not appear to be caused by dilution. Sorption and uptake may explain the reduction of conductivity.

5.8 Conclusions

Data from the LWDB and the OSUDWTS indicate that constructed wetlands are an effective treatment technology for treating dairy wastewater. Average reductions for COD, BOD, TS, TSS, TP, TKN, NH_3 and fecal coliforms were 45, 52, 27, 55, 42, 41, 37 and 80%, respectively. These reductions were less than the percent reductions reported for constructed wetlands treating domestic wastewater. However, the concentrations of COD, BOD, TS, TSS, TP, TKN, NH_3 and fecal coliforms found in domestic wastewaters are much lower than the concentrations found in livestock waste. The high loading rates at the OSUDWTS appear to cause oxygen limitations as indicated by the low DO of the influent and effluent. This may be reducing the treatment performance for BOD and nitrogen. Current volumetric and areal first-order plug flow models were fit to the data for the parameters. Table 5.27 summarizes the rate constants and the theta values for each parameter. Considering the simplicity of the volumetric and areal based models, both do an adequate job of explaining the observed reductions for all parameters ($\overline{R}^2$ = 0.65 and 0.67,

Table 5.27. Summary of volumetric and areal first-order constants for the Oregon State University Dairy Wetland Treatment System, Corvallis, OR.

	Volumetric Model				Areal Model			
Parameter	n[a]	$K_{20}(\sigma)$	$\bar{\theta}(\sigma)$	$\bar{R}^2(\sigma)$	C*[b]	$K_{20}(\sigma)$	$\bar{\theta}(\sigma)$	$\bar{R}^2(\sigma)$
COD	0.6	0.13 (0.02)	1.00 (0.01)	0.69 (0.06)	10	32 (5.9)	1.12 (0.02)	0.69 (0.10)
BOD	0.6	0.23 (0.04)	1.02 (0.01)	0.63 (0.11)	8.0	29 (5.2)	1.07 (0.01)	0.63 (0.16)
TS	0.6	0.09 (0.03)	1.00 (0.02)	0.72 (0.03)	20	10 (3.0)	1.03 (0.02)	0.75 (0.06)
TSS	0.6	0.29 (0.02)	1.01[c]	0.03 (0.02)	20	27 (2.6)	1.01[c]	0.01 (0.02)
TP	0.6	0.12 (0.02)	0.99 (0.01)	0.61 (0.05)	1.0	12 (0.9)	1.02 (0.01)	0.72 (0.07)
TKN	0.6	0.10 (0.01)	0.99 (0.00)	0.65 (0.05)	10	10 (1.4)	1.01 (0.00)	0.65 (0.14)
NH_3	0.6	0.13 (0.04)	1.02 (0.02)	0.70 (0.09)	3.0	14 (5.4)	1.05 (0.02)	0.73 (0.11)
Fecal Coliforms	0.6	0.48 (0.03)	1.01[c]	0.55 (0.11)	10	36 (2.4)	1.01[c]	0.50 (0.13)

[a] n held constant at 0.6 for all models

[b] C^* held constant at a specific value for each parameter

[c] held constant

respectively), with the exception of TSS ($\overline{R}^2$ = 0.03 and 0.01, respectively). No explanation could be found for the wide variability associated with TS, however, data from the LWDB showed similar variability for TSS.

Both models appear to be equally good for designing constructed wetlands for livestock waste given the current data available. One would expect the volumetric model to be better due to the inclusion of outlet flows, which should account for some of the variability. However, it appears that the addition of the outlet flows does not improve the predictive powers of the volumetric model (as indicated by the R^2) compared to the areal model. The averaging of the concentrations over a month or longer period in the areal model may have the same effect as including the outlet flows in the volumetric model.

One of the arguments for the volumetric model is that the rate constants are generally temperature dependent ($\theta > 1.05$), while the areal rate constants are not temperature dependent ($\theta = 1$) (Reed, 1995 draft). However, the average thetas found for the OSUDWTS were 1.01 (range = 0.99 to 1.02) and 1.04 (range = 1.01 to 1.07) for the volumetric and areal models. These findings conflict with the previous argument and indicate that temperature had little or no effect on the volumetric rate constant and a significant effect on the areal rate constants.

As mentioned above, both models appear to be equal at predicting removal rates based on the data from the OSUDWTS. However, the models are based on fundamentally different assumptions (Table 5.1). The volumetric model states that treatment is done by the surface attached microorganisms on plants and litter, and soil organisms, and therefore, increasing depth increases treatment performance. The areal model states that treatment is based on the surface area of the soil-water interface (soil organisms only), and therefore, increasing depth provides no additional treatment. Few data exist to support or refute either theory. This fundamental question must be addresses if wetlands are going to become a widely accepted tool for treating livestock waste. The "truth" may lie somewhere between the two models. Increased depth may not provide a proportional increase in treatment but it could provide some additional and necessary treatment for most parameters. However, until more data is collected one must choose which model to use and be aware of the assumptions inherent in each.

5.9 References

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6. Nitrogen Cycling in Constructed Wetlands: Theory

6.1 Abstract

Nitrogen cycling in constructed wetlands was reviewed. Data from the literature review were used to construct a conceptual model of nitrogen cycling. The conceptual model identified 20 state variables, 38 flow paths, and 12 forcing functions. Most flows could be modeled using first-order equations. Based on the rate constants, denitrification was clearly the most important removal mechanism. However, denitrification is closely coupled to nitrification, which is often limited by low dissolved oxygen in the wetland. This indicates that constructed wetlands designed for nitrogen removal must be optimized for maintaining high dissolved oxygen levels.

Keywords: simulation; wetland; nutrients

6.2 Introduction

For over 100 years wetlands have been conveniently used as wastewater discharge sites (Kadlec and Knight, 1996). It was noticed that wastewater improved in quality as it passed through a wetland. This recognition led to several studies during the early 1970's of wetlands used for municipal wastewater treatment (Odum et al, 1977; Ewel and Odum, 1984; Kadlec, 1983). The early studies showed wetlands to be a promising treatment system for municipal wastewater and their use has greatly increased since the mid 1980's (Kadlec and Knight, 1996). Today there are constructed wetlands treating all types of wastewaters. These include domestic wastewater, pulp and paper effluent, stormwater runoff, animal waste, agricultural nonpoint source pollution, sugar refinery waste, and landfill leachate. The wastewater characteristics of these sources are very different, as is the desired treatment. One common treatment goal is the removal of nitrogen because it has the potential for causing eutrophication in receiving waters and nitrate contamination in groundwater.

Much is known about nitrogen cycling in terrestrial and aquatic systems but less is known about nitrogen cycling in wetland systems. Most of the nitrogen pathways have been well documented but few attempts have been made to construct a complete nitrogen model for constructed wetlands. Currently, most constructed wetlands for nitrogen removal are designed using prior experience, best guesses, and empirical models (see Chapter 5). For constructed wetland design to improve, a better understanding is needed of the processes that occur within wetlands. One way to approach this is to develop a conceptual model of nitrogen cycling.

The objectives of this study are to:

1. review current literature related to nitrogen cycling in constructed wetlands;
2. develop a conceptual model of nitrogen cycling in wetlands;
3. identify the key processes involved in nitrogen removal; and
4. determine the data requirements to validate and calibrate a detailed nitrogen model.

6.3 Previous Wetland and Nitrogen Models

Compared to terrestrial and aquatic systems, the modeling of wetlands is a relatively new discipline (Mitsch et al., 1988). Most of the models constructed to date have been for specific wetlands and few generic models have been developed (Dørge, 1994). Almost all wetland models to date are composed of a hydrology and biological submodel. The hydrology submodels vary in the degree of complexity, from simple water budgets (Niswander and Mitsch, 1995; Mitsch and Reeder, 1991; and Mitsch, 1988) to more complex hydrodynamic transport models (Kadlec and Hammer, 1988). There are several early reviews of wetland models (Mitsch et al., 1982; Mitsch, 1983; Costanza and Sklar, 1985). The current state of wetland modeling is reviewed by Kadlec and Knight (1996).

Biological processes in wetlands have been primarily modeled using compartmental models (Dørge, 1994). Compartmental models show the transfer and storage of materials and energy in various wetland components. These models range from the very simple to the very complex but almost all fail to incorporate any spatial aspect (Kadlec and Hammer, 1988).

The main difficulty in building a spatial wetland model is the requirement of a large spatial and temporal data set, which is needed for model calibration and validation. The increase in collection of remotely sensed data and construction of geographical information systems (GIS) may provide many more opportunities to build spatial wetland models. A few attempts have been made at building spatial wetland models. Coastal wetland processes were simulated using both spatially explicit and implicit models (Costanza et al., 1990, Sklar et. al., 1985). The terrestrialization of fen ecosystems was simulated using a spatial and temporal model (Kooijman and Bakker, 1995).

6.4 Nitrogen Transformations and Processes

The major forms of nitrogen are organic nitrogen, ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), and dinitrogen gases (N_2O, N_2). The biological, chemical, and physical processes involved in nitrogen transformations in wetlands include:

1. nitrogen fixation;
2. mineralization/immobilization (ammonification);
3. ammonia volatilization;
4. nitrification;
5. denitrification;

6. plant and microbial uptake and release;
7. sedimentation and resuspension;
8. diffusion;
9. transport by ground water and surface water flow (hydrology and hydraulics);
10. atmospheric (chemical) deposition; and
11. leaching.

In constructed wetlands treating wastewater, nitrogen loadings are generally much higher than in natural systems. Nitrogen loadings in natural wetlands are generally around 1 or 2 g-N/m^2-yr with values up to 10 g-N/m^2-yr in riparian wetlands (Johnston, 1991). Constructed wetlands treating wastewater, on the other hand, usually receive nitrogen loadings well over 10 g-N/m^2-yr and in some cases over 1,000 g-N/m^2-yr (Kadlec and Knight, 1996). These high loading rates have several effects on the nitrogen cycle and allow for several simplifications to be made. Atmospheric deposition of nitrogen is generally less than 0.5 g-N/m^2-yr so it can be excluded from the model (Johnston, 1991). Constructed wetlands are almost always required to be sealed with either an impermeable membrane or heavily compacted clay layer which means nitrogen leaching and groundwater flows should not occur and can be left out of the model.

6.4.1 Nitrogen Fixation

Nitrogen fixation is the conversion of nitrogen gas (N_2) to organic nitrogen. Prokarotic bacteria and cyanobacteria containing the nitrogenase enzyme are able to carry out this process. Nitrogen fixing organisms can be free living, (asymbiotic) or associated with other organisms (symbiotic). Nitrogen fixers can be found on soil surfaces, in aquatic systems associated with plant roots or leaves, and in soils. The bacteria *Rhizobium* for example, is often associated with the root nodules of higher plants. The aquatic fern, *Azolla*, has a nitrogen fixing cyanobacteria associated with its leaves.

Nitrogen fixation is controlled by a wide variety of environmental and biological conditions (Buresh et al., 1980). In flooded soils, carbon availability and quality appears to limit nitrogen fixing by heterotrophic bacteria (Johnston, 1991). Nitrogenase is extremely oxygen sensitive and can only function at extremely low O_2 concentrations (Ogan, 1983). High levels of inorganic nitrogen, low light intensities (autotrophs only), high redox potentials, and pH level less than 5.0 or greater than 8.0 all inhibit nitrogen fixation (Ogan, 1983; Buresh et al., 1980). Temperature or growing season length may also effect nitrogen fixation (Johnston, 1991).

Contribution of nitrogen to wetlands by asymbiotic nitrogen fixers is reported to be between 0 and 6.7 g N/m^2-yr (Johnston, 1991). Studies have measured between 0.1 and 16.8 g N/m^2-yr added to natural wetlands by symbiotic nitrogen-fixing bacteria (Johnston, 1991). Studies of rice production often show an excess of nitrogen over what the soil or fertilizer can supply (Mikkelsen, 1987). This excess is assumed to be due to nitrogen fixation and can contribute 15-50 kg N/ha per crop (Koyama and App, 1979).

While nitrogen fixation may be significant in natural wetlands, it is insignificant in constructed wetlands. Constructed wetlands generally receive wastewaters containing high levels of inorganic nitrogen and nitrogen fixation is inhibited at very low levels of inorganic nitrogen.

6.4.2 Mineralization/Ammonification

Mineralization is the conversion of organic nitrogen forms to inorganic forms. Immobilization is the conversion of inorganic nitrogen forms to organic forms. Ammonification is the biological conversion of organic nitrogen to ammonium and is the most prevalent form of mineralization. Ammonia is released from organic material if the nitrogen content of the residue being decomposed exceeds the amount required by the microorganism. Under aerobic conditions, release of ammonium generally occurs when the carbon to nitrogen ratio (C:N) of the organic matter is less than 25:1. If the C:N ratio is greater than 25:1, nitrogen will be extracted from the mineral nitrogen pool (Paul and Clark, 1989). Under anaerobic conditions, ammonium is generally released from organic matter when the C:N ratio is less than 80:1 (Reddy and Patrick, 1984). The rates of ammonification, however, are much slower under anaerobic conditions compared to aerobic conditions. This is primarily due to the fact that under aerobic conditions there is a wide variety of general purpose heterotrophic bacteria and fungi that can carry out the decomposition. There are fewer anaerobic microflora capable of carrying out decomposition and the rate of decomposition is much slower because of the kinetics of anaerobic metabolism. The biochemical pathways of aerobic and anaerobic metabolism are also different, which explains the different C:N ratios (Paul and Clark, 1989).

In addition to the C:N ratio of the organic matter and oxygen state, the rate of ammonification is dependent on temperature, pH, chemical structure of the organic matter, available nutrients in the soil, and soil conditions (Reddy and Patrick, 1984). Under anaerobic conditions, the highest rate of ammonification is observed at 40 to 60° C, which is rarely observed in the field. However, it has been shown that the rate of ammonification under anaerobic conditions doubles with each 10° C increase in temperature (from 10° to 40° C) (Reddy and Patrick, 1984). The optimum pH range is near neutral (6.5 to 8.5)(Reddy and Patrick, 1984). The rate of decomposition of various organics is highly dependent on their structure (Paul and Clark, 1989). If soil nitrogen or other nutrients are limiting, than rates of decomposition can also be limited. Soil conditions, such as mineral and organic content can also effect ammonification. It is apparent that even modeling just nitrogen mineralization and immobilization can be very complex.

Some simplifications can be made when modeling a constructed wetland. It can be assumed that ammonification is only occurring in the sediments. The pH can be assumed to be near neutral because of the high loading of organics, which will tend to buffer the system (see Chapter 5). The nutrients in the soil will be assumed to be readily available because of the wastewater loading. The organic matter will be assumed to be of similar structure and be moderately decomposable with a decomposition rate k at 20° C = 0.14/day^{-1} and 0.02/day^{-1} for aerobic and anaerobic conditions (Paul and Clark, 1989). It can also be assumed that the soil in the wetland can be either aerobic or anaerobic. Thus, when modeling ammonification it will be necessary to have an equation for aerobic and anaerobic zones. Finally, it will be assumed that the decomposition will follow a first order decay curve.

$$dN_{org}/dt = -k_t \cdot t \qquad (6\text{-}1)$$

where, N_{org} = the organic nitrogen concentration in the sediments (g),

t = time (days), and

k_t = decomposition rate at a given temperature (day^{-1}).

One can use this equation to calculate the concentration of N_{org} at any time as follows:

$$N_{org_t} = N_{org_0} \cdot e^{-k_t \cdot t} \qquad (6\text{-}2)$$

where, N_{org_0} = concentration of N_{org_t} at beginning of time step.

Using this calculation one can also determine the net immobilization or ammonification based on the C:N ratio of the soil.

This is a simplified version of the mineralization and immobilization in soils but may be suitable for constructing a first cut model of N transformations in constructed wetlands. More complex models can be found in Paul and Clark (1989).

6.4.3 Ammonia Volatilization

Ammonia volatilization is a physicochemical process in which ammonium ion is in equilibrium with aqueous ammonia as indicated below:

$$NH_3(aq) + H_2O \leftrightarrow NH_4^+(aq)\ OH^- \qquad (6\text{-}3)$$

This reaction is pH dependent with high pH, greater than 8.5, favoring the ammonia (NH_3) form. Ammonia can be transferred to the atmosphere, depending on the partial pressures of the NH_3 in the water column and atmosphere. The amount of NH_3 lost depends on several additional factors including the initial concentration of ammonium, waste type, wind speed, temperature. At a pH of 7.0, ammonia concentrations are less than 1% of the ammonium concentration (Kadlec and Knight, 1996). Therefore ammonia volatilization is generally very low in constructed wetlands for most wastewaters. However, when ammonia concentrations are high, ammonia volatilization may be significant. Even though there is only a low concentration of ammonia in wetlands, it can be lost to the atmosphere and then the equilibrium will shift causing more ammonia to be formed. This process is used in municipal treatment plants and is optimized by increasing pH and using blowers to remove the gaseous ammonia. If ammonium concentrations are high and winds are sufficient to decrease the concentration of ammonia at the surface of the wetland, ammonia volatilization may be significant (Payne, 1996 draft).

6.4.4 Nitrification

Nitrification is the biological oxidation of ammonium (NH_4^+) to nitrate (NO_3^-). Nitrification is a two step process carried out by the aerobic chemoautotrophic bacteria of the genera *Nitrosomonas* (NH_4^+ to NO_2^-) and *Nitrobacter* (NO_2^- to NO_3^-) (Reddy and

Patrick, 1984). Nitrification can occur in the water column and at the oxidized soil-surface interfaces. The rate of nitrification in wetlands is controlled by temperature, pH, C source, microbial populations, and ammonium concentrations. Optimum rates of nitrification occur at temperatures from 30 to 35° C with the rate being greatly reduced at temperatures below 5° C and above 40° C (Reddy and Patrick, 1984). Near neutral pH, 6.6 to 8.0, is best for nitrification (Paul and Clark, 1989). If a carbon source is not available it can limit nitrification. The absence of microbial populations can also limit nitrification. Oxygen followed by ammonium are the most common limiting factors for nitrification in wetlands. Unless there is a readily available source of oxygen and NH_4^+, nitrification will not occur.

Many rate equations for nitrification in wetlands have been proposed. Among these are zero-order, first-order, and Monod population dynamics equations. Zero-order equations appear to be appropriate when ammonium concentrations and microbial populations are not limiting. The more complex Monod equation is useful if microbial populations are not at a maximum and one is interested in modeling the change in biomass of the microbial population. First-order equations are most appropriate when microbial populations are at a maximum and ammonium concentrations are limiting (Reddy and Patrick, 1984). Zero-order and first-order rate constants have been reported to be from 1.1 to 8.6 mg/ml-d and 0.003 to 9.00 d^{-1}, respectively (Reddy and Patrick, 1984).

For a constructed wetlands model, the rate constant will need to be adjusted for temperature. This could be done using the Arrhenius equation (5-3) defined in Chapter 5. pH can be assumed to be near neutral, as discussed in Chapter 5. The nutrients and carbon source can be assumed to be readily available due to the wastewater loading. The microbial populations should be at a maximum if the wetland is established. This leaves only the ammonium concentration to be modeled. If the ammonium concentration in the wastewater entering the wetland is known and than a first-order equation may be adequate. Nitrification should be calculated separately for both the surface water and aerobic soil layers.

It should also be noted that nitrate may be reduced to ammonia by various microorganisms. This is energetically favorable under anaerobic conditions and results in the formation of nitrous oxide and/or ammonia.

6.4.5 Denitrification

Denitrification is the conversion of nitrate to dinitrogen gas (N_2, N_2O) and is carried out by anaerobic bacteria. Oxygen, carbon availability, nitrate availability (nitrification rate), temperature, and pH all effect the rate of denitrification. Denitrification occurs only under anaerobic conditions at redox potentials of 350 to 100 mV. Constructed wetlands are excellent sites for denitrification because they usually have many anaerobic sites and an abundance of organic carbon. Denitrification is often the process that most scientists argue is the most important for removal of nitrogen in wetlands. Rates of denitrification from riparian wetlands are reported to average 6.8 g/m^2-yr and constructed wetlands can denitrify over 10 times as much (Johnston, 1991). Johnston (1991) and Reddy and Patrick (1984) both reported that the most import factors controlling denitrification are redox state, NO_3^- concentrations, and carbon (C) supply. If C or NO_3^- is not limiting then denitrification follows zero order kinetics (Hsieh and Coultas,1989; Reddy and Patrick, 1984). If C or NO_3^- is limiting then denitrification follows first order kinetics and finally if both NO_3^- and C are low than denitrification follows Michaelis-Menton kinetics (Hanson et al., 1994, Reddy and Patrick, 1984, Schipper et al., 1994) It is difficult to talk about denitrification without considering nitrification because of their close coupling. Rates of nitrification are often reported to be an order of magnitude lower than denitrification in both natural and constructed wetlands (Hseieh and Coultas, 1989; Johnston, 1991; Reddy and Patrick, 1984). This means that NO_3^- production is often the limiting step for denitrification. In addition, the diffusion of NO_3^- from the water column to the anaerobic soil sites can be limiting. Studies of constructed wetlands receiving wastewaters high in biochemical oxygen demand and high ammonia levels also have shown the nitrification step to be the limiting step for nitrogen removal (Kodmur et al., 1994; Stengel and Schultz-Hock, 1989). This indicates that it is of utmost importance to look at the coupled nitrification-denitrification processes, when studying N cycling in flooded soils (Hseigh and Coultas, 1989; Nielson, 1992). Temperature also affects denitrification rates and studies have shown that denitrification can occur at temperatures as low as 6-8°C, even though the optimum temperature is 60-75 °C (Reddy and Patrick, 1984; Stengel and Schultz-Hock, 1989).

As previously explained, the rate constant will have to be adjusted for temperature and could be done using the Arrhenius equation (5-3) defined in Chapter 5. As discussed above, the pH should be near close to neutral, the carbon source should not be limiting, and microbial populations should be at maximum levels in constructed wetlands. This leaves only the nitrate concentration and diffusion to be modeled. The nitrate concentration and diffusion into the anaerobic sediments can be assumed to be the limiting factors. Since only nitrate will be limiting, a first-order equation may be adequate. Diffusion of nitrate should be modeled and diffusion will be discussed below. Rate constants vary widely and would have to be determined for a specific site (Reddy and Patrick, 1984)

6.4.6 Plant and Microbial Uptake and Release

The flux of nitrogen into, within, and out of wetland plants is an extremely complex process that involves a number of pathways. Translocation of nutrients within plants, leaching of soluble nutrients, litterfall, senescence, root sloughing are a few of the pathways.

Each of these pathways vary for many different reasons. A few reasons are plant type, climatic region, harvest schedules, nitrogen loading rate, age of plants, and competition. In addition, calculations of density and reproduction must be considered. There are several approaches that can be taken for modeling plant growth and the associated uptake of nutrients. For a "first cut" model looking at nitrogen removal in a constructed wetland, it may be possible to make a great simplification and look at only the net uptake of nitrogen for plants over a given time period. For example, net retention of N was reported to be between 0.06 to 5.32 g/m^2-yr for natural wetlands (Johnston, 1991). This net uptake rate would probably not be too important in an overall nitrogen budget, considering that denitrification losses are estimated to be around 50 g/m^2-yr (Johnston, 1991). If plant material was harvested then N removal via plant uptake could be significant. However, harvesting is generally not economically feasible (Kadlec and Knight, 1996). Modeling the net uptake rates may be adequate if the biomass is monitored at the wetland being modeled and a gross estimate of removal by plants is made. Another confounding factor about plants is their ability to transport oxygen into the root zone. This may impact the nitrification process in the sediments but if the loading of the wastewater is high, then the oxygen delivered to the sediments would probably be preferentially used for aerobic respiration.

The simplifications discussed for plant uptake and release would limit the model's insight into the internal nitrogen dynamics in the wetland. However, the model would still be able to predict the nitrogen removal. The plant model could be expanded if it was found to limit the models overall utility. For a "first cut" model calculating the net uptake of nitrogen by plants in a wetland should be adequate. A zero-order uptake model could be used to predict removal of nitrogen from the inorganic N pool in the sediments. The equation would look like:

$$dN_{plant}/dt = k \qquad (6\text{-}4)$$

where, N_{plant} = the nitrogen content in the plants (g/m^2),

t = time (days), and

k = rate constant (g/m^2-day).

6.4.7 Sedimentation/Resuspension

Sedimentation and resuspension are physical processes that can either remove or add particulates, which may contain nitrogen. While both of these processes are primarily physical, organisms such as worms, fish, snails, insects, mammals, ducks, and alligators, can have very significant effects on resuspension (bioturbation). Activities of these organisms such as burrowing, grazing, and swimming can resuspend large quantities of particulates. While it is difficult to take the biological effects into account it is important to recognize that they exist.

Sedimentation (solids removal) is determined by the velocity of the water and the size and density of the particulates (see Chapter 5). Various models have been used to predict sedimentation in wetlands but most are not very accurate. (Kadlec and Knight, 1996). In order to determine the amount of nitrogen removed via sedimentation, it would be necessary to know the mass content or percent nitrogen content of the suspended solids and the settling rate. Use of a first- order model and the % nitrogen content may be adequate for a "first-cut" model.

6.4.8 Diffusion

There are several places where diffusion is important, such as: nitrate diffusion into the soils, N_2 diffusion from the sediments to the water column to the atmosphere, and ammonium diffusion to and from the sediments to the water column. There are several factors that effect each type of diffusion mentioned above. In a simple model, it can be assumed that anything that is denitrified will diffuse to the atmosphere quickly. In an ideal setting, the wastewater can be thought of as being uniformly mixed in the water column. If this assumed then only nitrate and ammonium diffusion to and from the waster column and sediments needs to be modeled. There are different approaches that can be used for modeling the diffusion but a good simple first approach is to assume a steady-state diffusion model as suggested by Bouldin et al. (1974):

$$V(dN/dt) = SD\cdot (N/L) \qquad (6\text{-}5)$$

where, dN/dt = the rate of change of nitrate concentration in the water with respect to time,

V = volume of water,

S = surface area of sediment water interface,

D = diffusion coefficient, and

L = thickness of the aerobic sediment layer.

This is a very simple approach and can also be used for ammonium diffusion. D should be measure for the wetland system to be modeled.

6.4.9 Hydrology and Hydraulics

The hydrology and hydraulics of a constructed wetland are two of the most important factors influencing N removal. A water budget is required for predicting mass loadings of N and a hydraulic model is needed to predict the transport of N. Hydrology and hydraulic models are discussed in detail in Chapters 3 and 4. N removal can only be modeled after hydrology and hydraulics are properly modeled.

6.5 Conclusions

An overall conceptual model based on the previous discussion was developed for a wetland (Fig 6.1). This model contains a state variable for rooted plants, plankton and floating plants, the water column, aerobic soil layer, and anaerobic soil layer. This is a total of five state variables, however, each state variable contains ammonia, ammonium ion, organic nitrogen, nitrite, nitrate, and soluble organic nitrogen. This means there are actually 20 state variables that must me modeled (Table 6.1). The model would also require 38 flow paths and rate constants. In addition, at least 12 forcing functions would have to be measured and used in the model (Table 6.1).

This level of detail would require an incredible amount of data collection to even get a snap shot picture of the various storage compartments. Even more data would be required to predict the fluxes and spatial distribution of N in a constructed wetland. Even if such a model is not practical to construct and validate, much can be learned from examining the conceptual model and reported rate constants for the various processes. The first obvious observation is that most of the nitrogen processes can be modeled using first-order models. This lends some support to the use of simple first-order models that use only inlet and outlet data (see Chapter 5). Secondly, denitrification has the fastest rate constant, indicating that it is the most important removal mechanism for N removal in constructed wetlands. However, denitrification is coupled with nitrification which is a much slower process and is often inhibited by low dissolved oxygen concentrations. This indicates that constructed wetland systems designed for nitrogen removal need to be optimized to have high dissolved oxygen levels and therefore high nitrification rates.

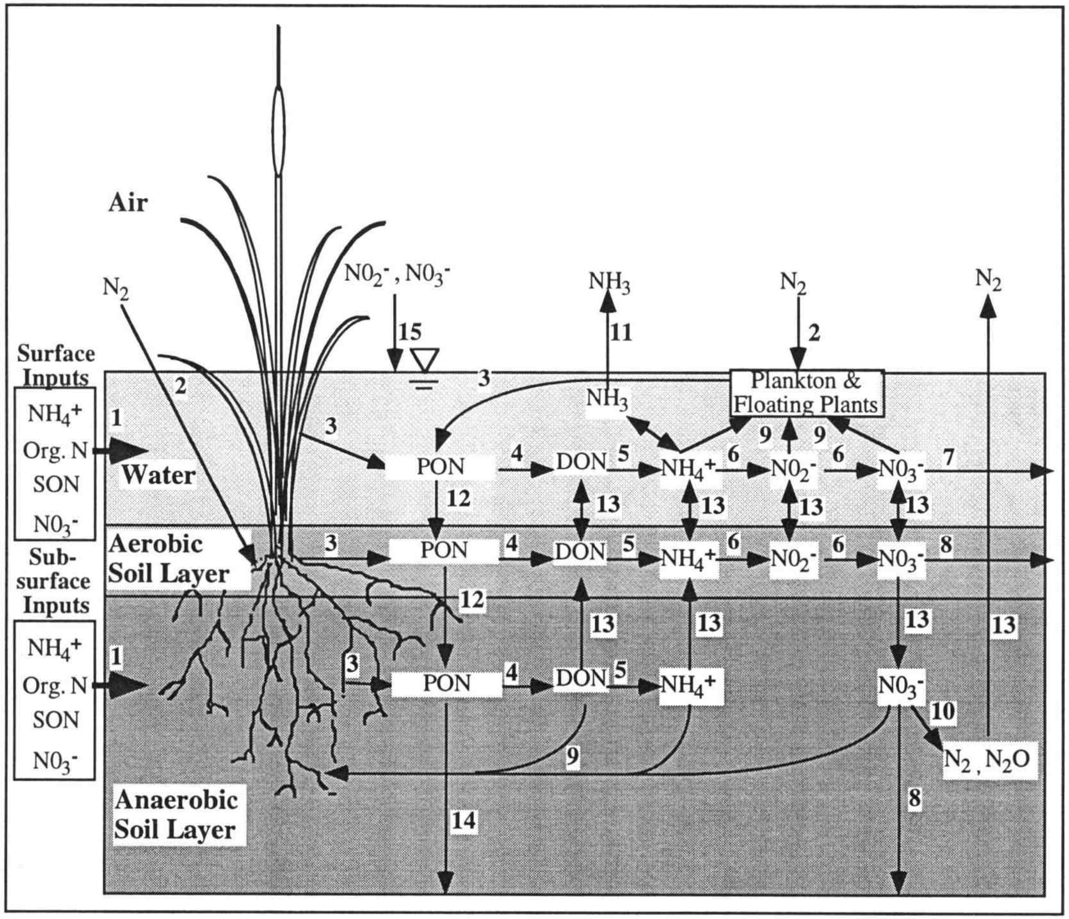

Figure 6.1. Conceptual model of nitrogen cycling in wetlands. PON = particulate organic nitrogen; DON = dissolved organic nitrogen. Pathways: 1 = inputs from surface and groundwater; 2 = nitrogen fixation; 3. = literfall and leaching; 4 = decomposition; 5 = mineralization/ammonification; 6 = nitrification; 7 = runoff; 8 = leaching; 9 = plant and microbial uptake; 10 =denitrification; 11 = ammonia volatilization; 12 = sedimentation; 13 = diffusion; 14 = long-term burial in sediments; and 15 = atmospheric (chemical) deposition.

Table 6.1. State variables and forcing functions for a nitrogen model for a wetland.

Symbol	Name	Values/Units
State Variables		
pN1	N in plants in the air and water column	g
pN2	N in plant roots in the aerobic soil layer	g
pN3	N in plant roots in the anaerobic soil layer	g
fpN1	N in floating plants and plankton	g
poN1	particulate organic N in water column	g
poN2	particulate organic N in aerobic soil layer	g
poN3	particulate organic N in anaerobic soil layer	g
doN1	dissolved organic N in water column	g
doN2	dissolved organic N in aerobic soil layer	g
doN3	dissolved organic N in anaerobic soil layer	g
NH_3	ammonia in water column	g
$NH_4 1$	ammonium ion in water column	g
$NH_4 2$	ammonium ion in aerobic soil layer	g
$NH_4 3$	ammonium ion in anaerobic soil layer	g
$NO_2 1$	nitrite in water column	g
$NO_2 2$	nitrite in aerobic soil layer	g
$NO_3 1$	nitrate in water column	g
$NO_3 2$	nitrate in aerobic soil layer	g
$NO_3 3$	nitrate in anaerobic soil layer	g
$N_2 3$	dinitrogen gas in anaerobic soil layer	g
Forcing Functions		
T(t)	temperature	°C
pH(t)	pH	pH
Ehw	redox potential in water column	mv
Eha	redox potential in aerobic soil layer layer	mv
Ehs	redox potential in anaerobic soil layer layer	mv
Q(t)	loading rate	L/day
oN(t)	concentration of organic N in influent	mg/L
soN(t)	concentration of soluble organic N in influent	mg/L
$NH_4(t)$	concentration of ammonium ion in influent	mg/L
$NO_2(t)$	concentration of NO_2^- in influent	mg/L
$NO_3(t)$	concentration of NO_3^- in influent	mg/L
C:N(t)	carbon:nitrogen ratio of influent	ratio

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7. Summary

It was the goal of this study to analyze and predict the treatment of dairy wastewater in a constructed wetland system. Treatment in constructed wetlands (CWs) is a function of several factors including hydrology, hydraulics, and kinetics. Therefore, all of these were studied at the Oregon State University Dairy Wetland Treatment System (OSUDWTS) prior to analysis of water quality data.

The first step was to predict evapotranspiration (ET) from the wetland system. It was shown that the Penman-Monteith equation was accurate for predicting wetland ET. Crop coefficients were developed for each of the dominant vegetation types. The ET loss from OSUDWTS was an average of 1.6 times the alfalfa reference ET. This ET loss was a small proportion of the annual water budget but had significant effects on seasonal detention time (DT). The wastewater hydraulic loading rate at OSUDWTS was high, which caused a short DT. If the wastewater hydraulic loading rate was decreased, ET would have a more significant effect on the overall water budget. In addition to predicting ET from CWs, ET calculations are needed by water resources managers for determining water use of natural wetlands. This study indicates that CWs can have high ET losses but additional research is needed to determine ET losses from different wetland types and vegetation types.

The second step was to evaluate the hydrology of the site and develop a complete water budget. It was found that seasonal patterns of rainfall and ET had a significant impact on detention time and treatment performance. The average monthly DT varied by as much as 18%. The seasonal patterns of rainfall and precipitation will vary from location to location and should be evaluated at each wetland site.

A water budget is needed for analyzing treatment performance, in addition, a measurement of the hydraulics is required. The hydraulics determine the flow path and contacting pattern. At OSUDWTS, it was found that the mean DT was an average of 0.58

times the theoretical DT. If the theoretical DT was used for predicting treatment or for developing rate constants, the results would be in serious error. It is very important that both the hydrology and hydraulics of a CW be determined, if data from the site are to be used to develop rate constants.

Once the hydrology and hydraulics have been measured, treatment performance and rate constants can be developed. Current design equations are based on either volumetric or areal first-order models. These equations require a minimum of inlet and outlet concentrations, temperature, wastewater hydraulic loading rates, and area of the wetland. The volumetric model also requires the porosity of the wetland (determined using a tracer study), the outlet flow (determined from a water budget), and the depth of the wetland. One can see even for these two simple models, a great deal of background data are required. Unfortunately, very few studies collect the required data and most studies make several assumptions regarding the hydrology and hydraulics. As shown in this study, ignoring the hydrology and hydraulics can result in significant errors in the predicted treatment and rate constants. For CWs to be accepted as a viable treatment technology, reliable design equations and criteria are required. These can only be developed with careful research that recognizes the importance of hydrology and hydraulics.

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